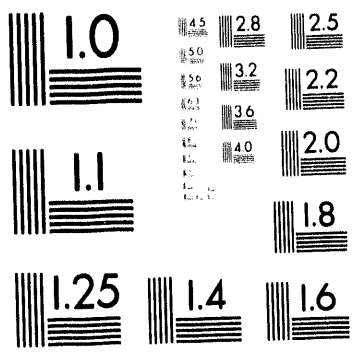
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Phase I Remedial Investigation Report for 200-BP-1 Operable Unit

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LIST OF ACRONYMS

ABS	dermal absorption factor
AMSL	above mean sea level
ARAR	applicable or relevant and appropriate requirement
ATSDR	Agency for Toxic Substances Disease Registry
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act of 1980
CFR	Code of Federal Regulations
CLP	Contract Laboratory Program
CMS	corrective measures study
CRDL	Contract Required Detection Limit
DCG	derived concentration guide
DHEW	U. S. Department of Health, Education and Welfare
DNR	Washington State Department of Natural Resources
DOE	U.S. Department of Energy
DOE-RL	United States Department of Energy, Richland Operations Office
DQO	Data Quality Objective
Ecology	Washington State Department of Ecology
EDMC	Environmental Data Management Center
EMI	electro magnetic inductance
EPA	U.S. Environmental Protection Agency
ERA	expedited response action
ERDA	United States Energy Research and Development Administration
FDM	Fugitive Dust Model
FS	feasibility study
HEAST	Health Effects Assessment Summary Tables
HEIS	Hanford Environmental Information System
HI	Hazard Index
HMS	Hanford Meteorological Station
HPGe	High purity Germanium
HQ	hazard quotient
HSBRAM	Hanford Site Baseline Risk Assessment Methodology
IARC	International Atomic Energy Agency
IAREC	Washington State University Irrigated Agriculture Research and Extension Center
IDL	Instrument Detection Limit
IRIS	Integrated Risk Information System
ITS	in-tank-solidification
LICR	lifetime incremental cancer risk
ICRP	International Commission on Radiological Protection
LOAELS	lowest observed adverse effect levels
MCL	maximum contaminant level
MCLG	maximum contaminant level goals
MTCA	Model Toxics Control Act
NAD	North American Datum
NCRP	National Council on Radiation Protection and Measurements
NCP	National Oil and Hazardous Substances Contingency Plan
NEPA	National Environmental Policy Act

LIST OF ACRONYMS (Cont.)

NGVD	National Geodetic Vertical Datum
NOAELS	no observed adverse effect levels
NOEL	no-observable-effect-level
NPL	National Priorities List
NPS	National Park Service
NRC	Nuclear Regulatory Commission
NTU	national turbidity unit
PAH	polyaromatic hydrocarbons
PARCC	precision, accuracy, representativeness, comparability and completeness
PCB	polychlorinated biphenyls
PNL	Pacific Northwest Laboratory
PSPL	Puget Sound Power and Light Company
QAPJP	Quality Assurance Project Plan
QC	Quality Control
RAGS	Risk Assessment Guidance for Superfund
RCRA	Resource Conservation and Recovery Act
RCW	Revised Code of Washington (State)
RfD	reference dose
RI	remedial investigation
RL	Richland Field Office
RLS	Radioactive Logging System
SAP	Sampling and Analysis Plan
SCS	Soil Conservation Service
SDG	Sample Delivery Group
SDWA	Safe Drinking Water Act
SF	slope factor
SMCL	secondary max
SQL	sample quantitation limit
SRC	Syracuse Research Corporation
STSC	Superfund Technical Support Center
TAL	target analyte list
TBC	to be considered
TBP	tributyl phosphate
TCL	target compound list
TOC	total organic carbon
UCL	upper confidence limit
UN	Unplanned Release
USC	United States Code
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
USWB	United States Weather Bureau
UTL	upper tolerance limit
VOA	volatile organic analysis
WAC	Washington Administrative Code
WDOW	Washington Department of Wildlife
WHC	Westinghouse Hanford Company
WPPS	Washington Public Power and Supply System
XRF	X-Ray fluorescence

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1.0 INTRODUCTION

The U.S. Department of Energy (DOE) Hanford Site, in Washington State is organized into numerically designated operational areas including the 100, 200, 300, 400, 600, and 1100 Areas. The U.S. Environmental Protection Agency (EPA), in November 1989 included the 200 Areas of the Hanford Site on the National Priority List (NPL) under the Comprehensive Environmental Response, Compensation and Liability Act of 1980 (CERCLA). The 200 Area NPL site is divided into eight source area groups largely corresponding to the major processing plants (e.g., B Plant) and two groundwater areas as defined by the Aggregate Area Management Strategy (Ecology, EPA, and DOE-RL 1991). Each source area group is further subdivided into one or more operable units based on waste disposal information, location, facility, type, and other site characteristics. The 200-BP-1 operable unit is one of several operable units located within the 200 East Area (Figure 1-1).

Inclusion on the NPL initiated the remedial investigation (RI) process for the 200-BP-1 operable unit. These efforts are being addressed through the Hanford Federal Facility Agreement and Consent Order (Ecology et al. 1989) which was negotiated and approved by the DOE, the EPA, and the State of Washington Department of Ecology (Ecology) in May 1989. This agreement, known as the Tri-Party Agreement, governs all CERCLA efforts at Hanford. In March of 1990, the Department of Energy, Richland Operations (DOE-RL) issued a Remedial Investigation/Feasibility Study (RI/FS) work plan (DOE-RL 1990a) for the 200-BP-1 operable unit. The work plan initiated the first phase of site characterization activities associated with the 200-BP-1 operable unit.

1.1 PURPOSE OF REPORT

The purpose of the 200-BP-1 operable unit RI is to gather and develop the necessary information to adequately understand the risks to human health and the environment posed by the site and to support the development and analysis of remedial alternatives during the FS. The RI analysis will, in turn, be used by Tri-Party Agreement signatories to make a risk-management-based selection of remedies for the releases of hazardous substances that have occurred from the 200-BP-1 operable unit.

In accordance with the Tri-Party Agreement, the 200-BP-1 operable unit RI and FS are conducted in a concurrent, interactively phased manner. Data collected and evaluated during the RI provide information needed to develop and analyze remedial alternatives during the FS, while the preliminary FS analyses provide a focus for further RI activities if needed. The goal of this first phase of the RI is to increase the initial understanding of the 200-BP-1 operable unit by characterizing the nature and extent of the threat to human health and the environment posed by releases of hazardous substances from the operable unit.

The RI report documents the findings of the Phase I RI to allow DOE-RL, EPA, and Ecology to assess the nature and extent of contamination, determine the need for further RI activities, and evaluate the FS. This report is consistent with the statutory requirements of CERCLA, the regulatory requirements of the NCP, and the Tri-Party Agreement.

Under the Tri-Party Agreement, a Phase I RI report is considered a secondary document. Therefore, this report is an interim interpretation of 200-BP-1 operable unit conditions. Review comments on this report will be considered in the development of a Phase II RI work plan and will be incorporated, along with Phase II RI findings, into a Phase II RI report, which is considered a primary document for decision making. In the event a Phase II RI is unnecessary to complete the FS, this Phase I RI report will become a primary document for decision making.

The 200-BP-1 operable unit originally included sources and off-site groundwater impacts. The 200-BP-1 Phase I Work Plan proposed investigations addressing data needs for both sources and groundwater. After implementation of the work plan, the concept, of evaluating risk and remedial actions for contaminated groundwater by aggregate area, became accepted. The change order, to the Tri-Party Agreement removed groundwater (off-site migration) from the 200-BP-1 operable unit. It was agreed during Unit Managers Meetings that RI data would be presented, the hydrogeologic system would be described and the extent of contamination would be identified within this 200-BP-1 Phase I RI Report. Fate and transport of contaminants within the groundwater systems and risk assessments associated with the groundwater pathway was agreed to be excluded from this report and deferred to the 200 East Groundwater Aggregate Area Study.

1.2 BACKGROUND

The Hanford Site, operated by the DOE, occupies approximately 1450 km^2 (560 mi^2) of southeastern part of Washington State north of the confluence of the Columbia and Yakima rivers. Since 1943, the Hanford Site has been used for reactor operations, reprocessing of spent fuel and management of radioactive waste. The historical mission was plutonium production for defense purposes; plutonium was produced in reactors located in the 100 Areas via the irradiation of uranium fuel and subsequently separated from other substituents in the irradiated fuel at separations plants located in the 200 Area. In recent years, the emphasis of the mission has changed from production of special nuclear materials to waste management and environmental restoration.

The 200-BP-1 operable unit is located within the north-central portion of the 200 East Area (see Figure 1-2). The operable unit includes 14 waste management units (10 inactive cribs and 4 unplanned releases) and encompasses approximately 10 ha (25 ac) with the majority of the waste management concentrated in a 1.6 ha (4 ac) region at the eastern end of the operable unit. The 200-BP-1 waste disposal activities were associated with the management of waste from U Plant uranium reclamation operations and waste storage condensate from the adjacent 241-BY Tank Farm (see Sections 3.1 and 4.1 for more information).

The exact concentration and quantity of radionuclides and contaminants of concern remaining within the operable unit is uncertain. Historical records indicate that seven cribs (cribs 216-B-43 through -49) received an estimated 33,840,000 l (8,940,000 gl) of tributyl phosphate supernatant waste from 1954-55, 2 cribs (cribs 216-B-50 and 216-B-57) received an estimated 139,200,000 l (36,777,000 gl) of in-tank-solidification (ITS) condensate from 1965-74, and one crib was constructed (crib 216-B-61) but has no documentation of past disposal operations. More details on waste inventories to cribs are provided in Chapter 4.1.

1.3 REPORT ORGANIZATION

The 200-BP-1 operable unit Phase I RI report is organized in a format similar to that recommended by EPA (EPA 1988a). In addition to this introduction, the RI report consists of the following 7 chapters and appendices:

- Chapter 2.0, Phase I Data Collection Activities summarizes actual data collection activities associated with the "RI/FS Work Plan for the 200-BP-1 Operable Unit, Hanford Site, Richland, Washington" (DOE/RL 88-32, Rev.1).

- Chapter 3.0, Physical Characteristics of the 200-BP-1 operable unit provide a description of relevant physical characteristics.

- Chapter 4.0, Nature and Extent of 200-BP-1 Operable Unit Contamination summarizes contaminants of concern based on empirically-determined chemical and contaminant characteristics for both groundwater and source areas.

- Chapter 5.0, Contaminant Fate and Transport Analysis represents analyses of Sections 3.0 and 4.0 to develop interpretations of environmental fate and transport of operable unit contaminants. Transport modeling is applied in this section to estimate current and future concentrations in each environmental medium.

- Chapter 6.0, Baseline Risk Assessment, estimates the actual human health and environmental threats posed by hazardous substances released from the 200-BP-1 operable unit.

- Chapter 7.0, Summary and Conclusion, summarizes all Phase I RI activities and provides recommendations for the FS and additional RI activities.

- Chapter 8.0, References, provides a list of cited documents within the body of the Phase I RI report.

- Appendices are used to present letters and memoranda cited, concise summaries of validated/unvalidated data and detailed technical analyses needed to confirm the findings contained within the text. To avoid redundancy, such information is incorporated by reference, rather than appended, whenever it is published and readily available to data users.

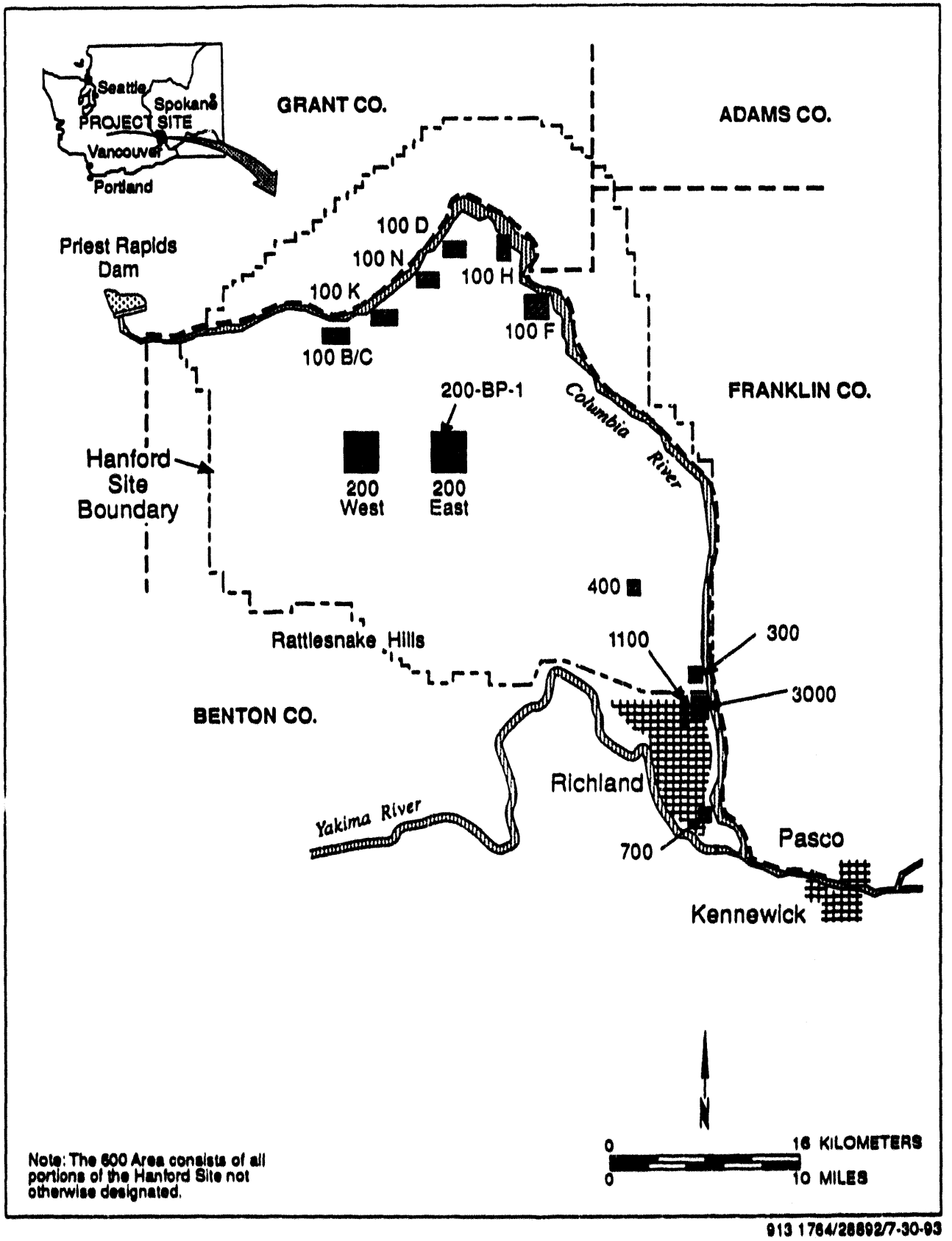


Figure 1-1. Hanford Site Map.

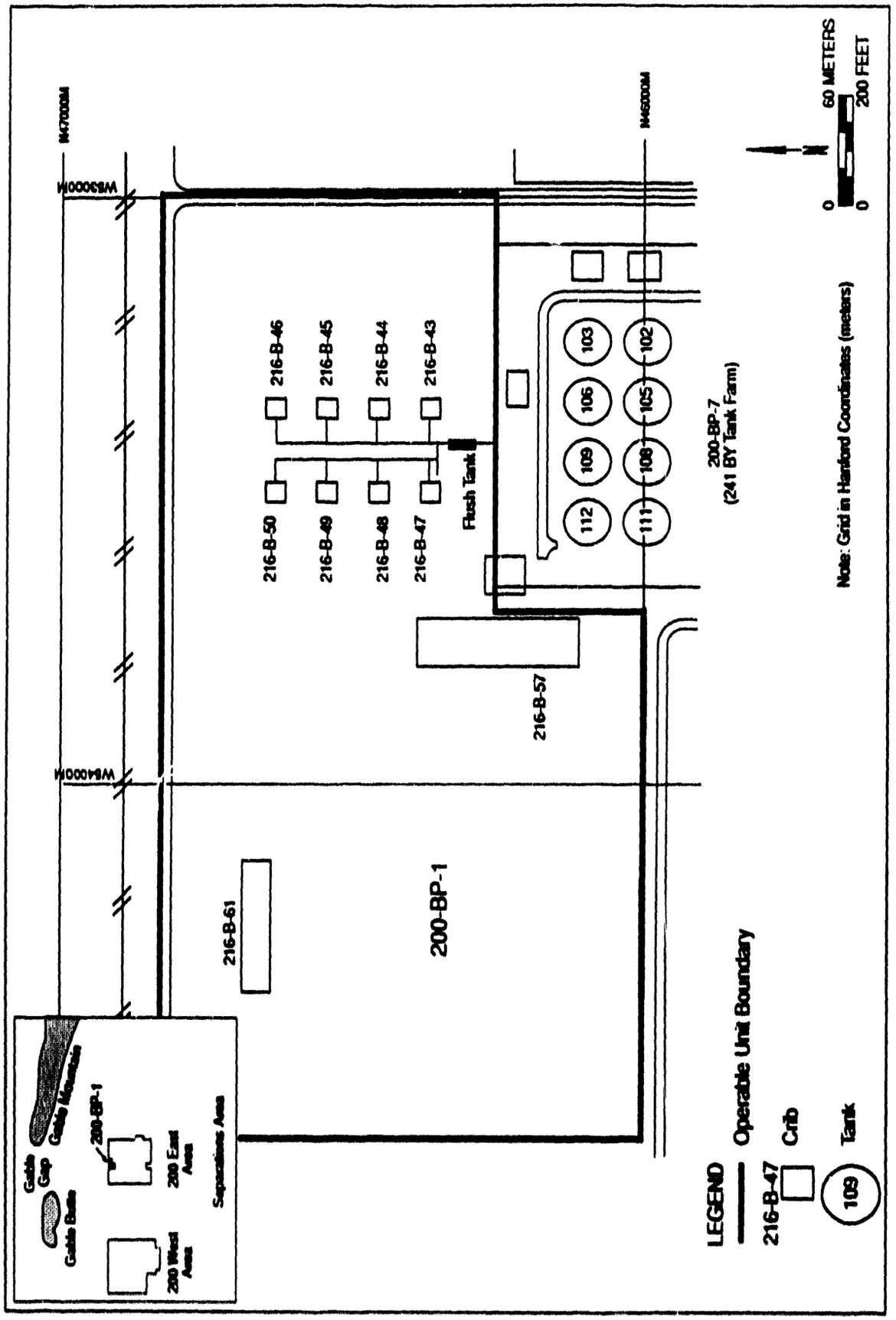


Figure 1-2. The 200-BP-1 Operable Unit.

2.0 PHASE I DATA COLLECTION ACTIVITIES

The first phase of the 200-BP-1 operable unit RI was conducted in accordance with the 200-BP-1 operable unit RI/FS work plan (DOE-RL 1990a). This chapter provides a summary of the various data collection activities that were undertaken during the course of Phase I of the RI. Interpretations of these data are provided in subsequent chapters of this report. Table 2-1 summarizes each data collection activity and provides the respective samples and analyses conducted for Tasks 2 through 12 as described in the 200-BP-1 RI/FS Work Plan.

Phase I RI data collection activities are presented below by the following environmental-medium- or environmental-discipline-specific task categories:

- Source and vadose zone (Tasks 2, 4, and 10)
- Surface soils and topography (Tasks 3 and 8)
- Hydrogeology and geology (Tasks 5, 6, 7, 11 and 12)
- Ecology (Task 9).

2.1 SOURCE AND VADOSE ZONE INVESTIGATION (TASKS 2, 4 AND 10)

This subsection describes data collection activities which were associated with characterization of the contaminant source materials and the underlying vadose zone sediments. As such, the following tasks of the 200-BP-1 operable unit work plan will be addressed herein:

- Source and vadose zone sampling and analysis (Tasks 2 and 4)
- Column leach testing (Task 10).

The purpose of the source and vadose zone investigations was to determine the extent of contamination in the crib source materials and underlying unsaturated soils. The crib gravels and the sediments lying immediately beneath the ten 200-BP-1 operable unit cribs were sampled to determine contaminant levels associated with each crib. The goals of the data gathering effort listed in the work plan for the source and vadose zone characterization consisted of the following:

- Identification of all waste constituents remaining in the cribs
- Collection of detailed geologic and stratigraphic information to assess the possibility of perching and lateral migration of infiltrating liquid waste effluent
- Determination of waste constituent concentration profiles with depth to evaluate vertical migration and assess future impacts to groundwater
- Collection of representative vadose zone samples for laboratory testing. Testing was to include column leach tests, physical property tests, sorption tests, and

potential bench-scale treatment tests (if determined to be necessary in the future).

2.1.1 Drilling and Soil Sampling (Tasks 2 and 4)

Drilling and sampling activities associated with the source and vadose zone investigations were conducted in accordance with Tasks 2 and 4 of the 200-BP-1 Sampling and Analysis Plan (SAP). Cable-tool drilling methods were used for drilling and to collect soil samples for chemical and physical analysis. Soils were extracted from each borehole via a split-tube or drive barrel sampler. Radioactive samples were prepared in a field glovebox and shipped to an appropriate physical or analytical laboratory, depending on radiation levels. Figure 2-1 depicts the locations of the Task 2 and 4 boreholes. Table 2-2 summarizes the relevant completion data for each Task 2 and 4 borehole.

As defined under Task 2, three boreholes were drilled through each crib, except for crib 216-B-61. One borehole was drilled in the 216-B-61 crib to confirm that no waste effluent was discharged to the crib, as suspected (See Section 4.1). Each Task 2 borehole in the 216-B-43 through 216-B-50 cribs was drilled to a depth of approximately 3 m (10 ft) below the gravel infiltration layer (approximately 9 m [30 ft] below ground surface). The boreholes were drilled in a triangular array, as shown in Figure 2-1, so as not to disrupt the existing crib piping and structure. Four samples were collected for chemical analysis from each borehole, as depicted schematically in Figure 2-2.

Task 2 boreholes in the 216-B-57 crib were drilled to a depth of approximately 15.2 m (50 ft) below ground surface. Samples were collected at the same intervals as for the 216-B-43 through 216-B-50 cribs. Boreholes were drilled deeper than originally planned at the 216-B-57 crib due to the extended vertical depth of contamination.

As defined under Task 4, three boreholes were drilled as part of the 200-BP-1 operable unit phase I RI. One was drilled through each of the 216-B-43, 216-B-49, and 216-B-57 cribs (299-E33-296, 299-E33-302, and 299-E33-304, respectively) (Figure 2-1). Task 4 boreholes represented extensions of selected Task 2 boreholes in which drilling continued to a total depth of approximately 70 m (230 ft) for each borehole. Chemical samples from the Task 4 boreholes were collected every 7.6 m (25 ft). Continuous split tube samples were collected from the 299-E33-304 borehole to determine stratigraphic and physical property data at depth beneath the 200-BP-1 operable unit.

Due to the uncertainties regarding the nature of chemicals discharged to the cribs, samples from the Task 2 and 4 boreholes were analyzed for all CERCLA TCL and TAL constituents, major anions, bismuth, cyanide (free, complexed and total), and for major radioisotopes. Table 2-1 lists the number of samples analyzed for each constituent. Evaluation of the chemical analytical data for Task 2 and 4 samples is presented in Chapter 4.

Selected Task 2 and 4 samples were physically tested. Physical tests included bulk density, moisture content, grain size, moisture retention, saturated and unsaturated hydraulic conductivity, specific gravity, calcium carbonate, and porosity.

Table 2-2 summarizes relevant borehole completion data, including borehole number, crib, location coordinates, total depth, and numbers of samples collected for physical and

chemical testing. Boring logs, physical testing results, and detailed descriptions of the drilling activities are presented in Hoffman (1992) and summarized in Section 3.4. Moisture retention data and unsaturated hydraulic conductivity measurements are presented in Connelly et al. (1992) and summarized in Chapter 5.2.3. A comprehensive listing of all Task 2 and 4 chemical samples, with Hanford Environmental Information System (HEIS) numbers, sample depths, and analyses performed is presented in Appendix A-1. Analytical results are summarized in Appendix A-2. Quality assurance results associated with Tasks 2 and 4 samples are presented in Appendix A-3.

Selected Task 2 and 4 boreholes were geophysically logged using a spectral gamma-ray tool during the construction of each borehole, allowing for the collection of continuous and semi-quantitative radioisotopic data. Several Task 6 boreholes and a number of pre-existing monitoring wells were also logged with the geophysical tool. Table 2-4 lists the boreholes where the spectral gamma-ray geophysical technique was employed. Due to the high activity levels directly underneath the crib gravels, the tool became saturated. The results of the spectral gamma logging are presented in Price (1992) and are discussed in Section 4.4.

Upon completion of Task 2 and 4 boreholes, each borehole was abandoned according to WAC 173-160 by removing steel casings and backfilling with granular bentonite. A permanent marker was place at each drilling site and subsequently surveyed for elevation and horizontal coordinates. Survey data are presented in Table 2-2.

2.1.2 Background Data Collection

Background soil quality was characterized from samples collected from Task 2 borehole 299-E33-307 (Figure 2-1) and from Task 6 boreholes 699-52-57 and 699-55-55, which were drilled north of the 200-BP-1 operable unit in the 600 Area. Borehole 299-E33-307 was drilled through the 216-B-61 crib which was believed to have never been used for waste disposal. Results of the background soil analyses are summarized and evaluated in Chapter 4. All Task 2 and 4 analytical data are presented in Appendix A. Task 6 soil analytical data is discussed below in Section 2.3.

2.1.3 Column Leach Testing (Task 10)

Task 10 of the work plan consisted of the performance of a series of column leaching laboratory experiments using samples collected from the Task 4 drilling activity. The purpose of the tests was to assess potential contaminant fluxes which could occur through the vadose zone from infiltrating water. Column leach samples were collected from boreholes 299-E33-296, 299-E33-302, and 299-E33-304. Tests were completed as described in the column leach test plan (Crane 1992). A data package for the column leach testing is presented in Appendix B. Analytical data associated with Task 10 are included in Gillespie (1992).

2.2 SURFACE INVESTIGATION (TASKS 3 AND 8)

This subsection presents a description of data collection activities which were associated with the surface environment of the 200-BP-1 operable unit. As such, the following tasks of the 200-BP-1 operable unit work plan are addressed:

- Surface and near-surface soil sampling (Task 3)
- Preparation of a detailed topographic map of the 200-BP-1 operable unit (Task 8).

The purpose of the near surface soil sampling (Task 3) was to assess contaminant levels in soils at or near the surface. Several unplanned releases (UN-200-E-9, UN-200-E-63, UN-200-E-110, and UN-200-E-89) are associated with the 200-BP-1 operable unit, but the location, extent, and concentration of contaminants were uncertain. In addition several underground distribution pipelines were tested to locate possible leaks. The goals of the data gathering effort stated in the work plan for Task 3 are the following:

- Delineate surface contamination due to unplanned releases (as modified per work plan change request 13, approved March 22, 1991)
- Evaluate leak-detection technologies for underground effluent distribution lines
- Locate possible soil contamination due to pipeline failures

A topographic map (Task 8) was needed in order to provide a suitable base sheet for use in site characterization, evaluation of remedial alternatives, and engineering design.

2.2.1 Surface Scintillation Survey (Task 3)

The ground surface of the entire operable unit was surveyed using alpha and beta/gamma scintillation detectors. The crib area was surveyed by hand held instruments. The remaining areas in the operable unit were surveyed using a tractor mounted scintillation tool. Areas with either alpha or beta/gamma readings above background were flagged for further investigation. Each area was geodetically surveyed for plotting on the base topographic map. The results of the scintillation survey are presented in WHC (1989) and the area of surface contamination is depicted in Figure 2-3. Additional details on the "hot spot" locations are provided in WHC Drawing H-2-78769, Rev. 0. Most of the surface of the 200-BP-1 operable unit had surface radioactive contamination present. The majority of contamination may have been derived from wind blown particulate matter (Hayward 1992). Possible sources are considered to be the unplanned releases from the operable unit and adjacent tank farm areas.

2.2.2 Surface Sampling and Stabilization

To reduce the spread of surface contamination identified in the surface scintillation survey, surface stabilization activities were implemented in the summer and fall of 1991 (Hayward 1992). Prior to commencing stabilization activities, near surface soil samples were collected at 26 locations throughout the 200-BP-1 operable unit. Samples were

collected at areas of near surface contamination, approximate locations of unplanned releases, near the flush tank, and on the west side of the operable unit for background data. Each sample location was geodetically surveyed for plotting on the base topographic map. Figure 2-4 depicts the locations of the Task 3 soil samples. Soil samples were analyzed for radioisotopes, nitrate, phosphate, sulfate, bismuth, selenium, total cyanide and free cyanide. Analytical results for the Task 3 soil sampling activity are evaluated in Chapter 4 and summarized in Appendix C.

Surface stabilization activities consisted of removing up to 15.2 cm (6 in.) of material on the eastern portion of the operable unit and placing it over the crib areas. The crib areas were then covered with approximately 46 to 61 cm (18 to 24 in.) of clean material. The area was then resurveyed with scintillation detectors to verify removal of all surface contamination. The operable unit was reposted with only subsurface radiological contamination around the crib area (Figure 2-3). More information regarding the stabilization activities is presented in Chapter 3. Hayward (1992) consists of a detailed summary of the stabilization activities.

2.2.3 Pipeline Integrity Testing

An engineering study was undertaken, as called for in Task 3 of the work plan, to evaluate pipe penetration and leak detection techniques that would be applicable for testing underground waste effluent pipelines. Figure 2-4 shows the location of underground pipelines in the 200-BP-1 operable unit (additional details are provided in WHC Drawing H-2-78769, Rev. 0). The results of the evaluation are presented in Hayward (1990). A hot tap technique was determined to be the preferred method for accessing the pipelines, and a helium tracer gas method was chosen for leak detection testing.

Pipelines that were evaluated included two 10.2-cm (4-in.) pipelines running east-west between the tank farm and the operable unit, and a 5.1-cm (2-in.) pipeline running from the tank farm to the flush tank within the operable unit. The pipeline running to the 216-B-61 crib was not evaluated since it was determined that the crib did not receive waste.

Prior to testing the pipelines in the field, a nondestructive examination was completed to determine the presence of liquids. An ultrasonic thickness measurement (WHC procedure NDT-UT-900, rev. 2) was performed on each pipeline. The nondestructive testing indicated that the 5.1 cm (2 in.) line and the south 10.2 cm (4 in.) line (Line 2805-E-1) were filled with liquid, while the north pipeline (Line 2805-E-2) was empty. Initial helium tests on the empty pipeline indicated no leaks. No further testing was performed on pipelines that contained liquid. A final report of all field activities was not available at the time of this report.

2.2.4 Topographic Mapping (Task 8)

As defined under Task 8, a topographic map of the operable unit was prepared. A field survey was completed in the summer of 1989 to collect data for map preparation. As the interim stabilization activities at the operable unit occurred in 1991, the map is representative of conditions prior to interim stabilization. Interim stabilization is described in Section 2.2.2 and in Hayward (1992).

The map was prepared to contour intervals of 0.5 m (1.6 ft). Official copies of the base map are located in the Westinghouse Hanford Engineering files as drawing number H-2-78769, rev. 0. The map is certified by a Washington State Professional Land Surveyor. A topographic map of the operable unit, based on this drawing, is depicted in Figure 2-4.

2.3 HYDROLOGIC AND GEOLOGIC INVESTIGATION (TASKS 5, 6, 7, 11, AND 12)

The hydrologic and geologic investigation activities were completed to determine site-specific geologic and hydrologic condition associated with the 200-BP-1 operable unit. Data collection activities were associated with Tasks 5, 6, 7, 11, and 12 of the 200-BP-1 work plan.

2.3.1 Seismic Refraction Survey (Task 5)

Task 5 of the work plan consisted of the planned performance of a seismic refraction survey in the area immediately to the north of the 200-BP-1 operable unit. The purpose of the seismic work was to more accurately define the surface elevation of the uppermost basalt stratum in the general area of contaminant plume migration. Emphasis was to be placed on the identification of possible paleochannels in the basalt which could influence the migration of contaminant plumes, and on the further delineation of a known erosional "window" in the basalt (in the vicinity of well 699-53-55) which could enhance hydraulic communication between the unconfined and confined aquifers. Procedures for conducting the survey and potential impacts to the nearby tank farms were to be evaluated in a field test. The primary purpose of the field test was to evaluate the feasibility of the technique in providing useful information for the RI. The results of the field test were reported in Buckmaster (1993a).

The energy source for the seismic survey consisted of 10-gauge shotgun shells instead of the proposed kinepak two-component explosives. These were used in the field test performed to assess the surveys usefulness. As reported in Buckmaster (1993a), the shotgun shells proved to be inadequate for use in the survey. Insufficient percussive energy was created by the shells to produce definitive seismic reflections from the basalt surface, thereby leading to inconclusive results. The bedrock surface could not be clearly delineated. Any additional seismic surveys was deferred to the 200 East Aggregate Area Groundwater Study.

2.3.2 Installation of Groundwater Monitoring Wells (Task 6)

Task 6 of the work plan involved the installation of 10 groundwater monitoring wells. Seven wells were drilled to monitor the unconfined aquifer and three were drilled to monitor the Rattlesnake Ridge confined aquifer. The goals of the Task 6 well installation activities were the following:

- Delineate contaminant plumes in the unconfined aquifer that are associated with sources in the 200-BP-1 operable unit.
- Determine the potential future movement of contaminants emanating from 200-BP-1.

- Evaluate the impacts or potential impacts to the Rattlesnake Ridge aquifer.
- Establish cluster well systems (monitoring wells in both the confined and unconfined aquifers) to evaluate the hydraulic interconnectivity of the aquifers.
- Determine subsurface stratigraphy.
- Determine vadose zone soil properties.
- Collect soil samples to determine background chemical concentrations for comparison with Task 4 vadose sampling.

2.3.2.1 Monitoring Well Locations. Groundwater monitoring wells were located based on existing aquifer characteristics in the region. As discussed in Section 2.3.1, an evaluation of seismic data completed under Task 5 was inconclusive in delineating potential erosional windows or paleochannels in the upper most basalt member. The monitoring wells were therefore, located based on existing hydrogeologic data. Figure 2-5 depicts the location of each monitoring well installed under Task 6. Table 2-3 summarizes relevant well construction data.

2.3.2.2 Well Installation Data Collection Activities. Conventional cable-tool drilling techniques were used to install the 10 new groundwater monitoring wells. Soil samples were collected every 1.5 m (5 ft) or at changes in lithology observed during the drilling. Selected samples were submitted for physical property analyses. Results of these analyses are presented in Hoffmann (1992). Borehole geologic logs and well installation details are presented in Hoffmann (1992). An evaluation of this data is provided in Chapter 3 and in Hoffmann et al. (1992).

Soil samples for chemical analysis were collected from wells 699-52-57, 699-55-55, 299-E33-38, and 299-E33-40. Soil samples were analyzed for total organic carbon (TOC), major anions, bismuth, selenium, total cyanide, free cyanide and radioisotopes. Results of the soil chemical analyses are presented in Chapter 4 and summarized in Appendix D. Table 2-1 lists the number of samples analyzed for each constituent.

Each of the 600 Area wells were geophysically logged with a gross-gamma tool. Results are presented in Hoffmann (1992). The three 200 Area wells were logged with a spectral gamma tool as per the work plan change request 3 (approved May 1, 1990). Results are presented in Price (1992) and discussed in Chapter 4.

2.3.3 Hydrologic Investigation

Two tasks in the work plan were associated with determining site specific hydrologic conditions. Task 7 involved the performance of periodic groundwater sampling throughout a 200-BP-1 groundwater monitoring network. Task 11 activities were associated with the measurement of hydraulic properties of the unconfined and Rattlesnake Ridge confined aquifers.

2.3.3.1 Groundwater Sampling and Analysis (Task 7). The goals of Task 7 were the following:

- Evaluate existing groundwater monitoring wells to determine acceptability for future monitoring.

- Perform quarterly and semiannual groundwater sampling and analysis for the 200-BP-1 operable unit groundwater monitoring well network.

- Develop analytical methods for CN specification, and to reduce the detection limit for Ru-106.

2.3.3.1.1 Evaluation of 200-BP-1 Operable Unit Groundwater Network. As part of Task 7, an evaluation of the existing monitoring wells was performed to determine well condition, suitability for use in the monitoring network, and possible well modification (remediation) needs. Depending on the condition of each well, remediation activities were performed. These included camera surveys, scrubbing, screen interval modifications, and installation of surface seals and/or surface pad and posts. The results of this work, including current as-built drawings for each well, are presented in Buckmaster (1993b).

Wells sampled under the Task 7 activities are listed in Table 2-5 and depicted in Figure 2-6. A total of 44 wells are included, 37 of which monitor the uppermost aquifer and 7 of which monitor the Rattlesnake Ridge confined aquifer.

2.3.3.1.2 Groundwater Sampling and Analysis. Quarterly groundwater sampling was initiated in January of 1991. Monitoring wells were sampled quarterly for the first year, then sampled semiannually until completion of the groundwater FS. Due to the uncertainties regarding the types of contaminants present in groundwater and the quality of historical data, the first sampling event included analyses for all CERCLA TCL and TAL constituents, cyanide, major radioisotopes, and constituents through wet chemistry. Constituents analyzed in groundwater are presented in Appendix E.

After the first sampling event, which confirmed that organic compounds were not present in the groundwater, the TCL constituents (volatiles, semivolatiles and pesticides) were eliminated from the suite of analyses performed.

Field parameters were collected at each well during sampling. Parameters measured included pH, temperature, electric conductivity, turbidity, and dissolved oxygen. Field parameters are presented in Appendix E.

During the first five sampling events, a limited number of wells were not sampled due to well remediation activities or pump malfunctions. Because of these interruptions, from 36 to 44 wells were sampled during each sampling event . Table 2-5 indicates the wells sampled during each quarter and the analyses performed.

The evaluation and summary of the groundwater sampling and analysis results obtained under Task 7 are presented in Chapter 4 in a series of contaminant plume maps. A complete list of all groundwater analytical data is found in Appendix E.

2.3.3.1.3 Cyanide Speciation and Ruthenium-106 Analyses. Under Task 7 of the work plan, PNL was contracted to develop analytical methods for the following two measurements:

- Analysis of cyanide speciation in groundwater samples, i.e., an analytical procedure for the determination of total, free and complexed forms of cyanide
- Analysis of Ru-106 in groundwater to a new, reduced detection limit of 3 pCi/l.

Results of the work performed to develop these analytical procedures are presented in WHC (1989).

Prior to initiating the first round of groundwater sampling, an evaluation of existing groundwater data was completed to determine the presence of Ru-106 (DOE-RL 1990a). Based on this evaluation, Ru-106 analysis was performed for selected groundwater samples for the initial round of sampling. Ru-106 was not detected in these samples (maximum detection limit was 4.6 pCi/l), and Ru-106 analysis was eliminated from future sampling events.

For cyanide speciation, analytical methods were developed by PNL to measure free and total cyanide concentrations, as presented in WHC (1989). The concentration of complexed cyanide is calculated as the difference between free and total concentration values.

Analyses for cyanide speciation, i.e., total and free (and the resulting complexed cyanide component) were performed on all samples for the first two sampling events. Total cyanide analysis was performed by two separate laboratories (PNL and Weston). Free cyanide analysis was performed by PNL. Evaluation of preliminary data indicated elevated levels of CN in only a few monitoring wells. Following the second sampling round, analysis for free cyanide was reduced to selected wells which exhibited elevated levels and was then eliminated from future sampling events. Total cyanide analysis was continued on all wells. Cyanide results are summarized in Appendix E and evaluated in Chapter 4.0.

2.3.3.2 Aquifer Testing (Task 11). The purpose of the aquifer tests (Task 11) was to obtain information on the hydraulic properties of the unconfined and Rattlesnake Ridge aquifers. The selection of wells and the hydraulic test method used for each well was based on well construction and groundwater quality in accordance with the work plan. Slug tests were performed on all Task 6 monitoring wells. Results of the slug tests are presented in Hoffman (1992). Drawdown/recovery testing was performed at wells 699-49-57B, 699-52-54, 699-52-57, and 699-53-55C. A presentation of the results of this testing can be found in Swanson (1992). Data obtained from these tests include transmissivity, hydraulic conductivity, and specific yield. Results of the aquifer testing are discussed in Chapter 3.0.

2.3.3.3 Sorption Testing. Sorption testing was an optional test included under Task 12 of the work plan. The purpose of the tests was to determine contaminant sorption coefficients (K_d) in the unconfined aquifer. A literature search was performed to obtain available data pertaining to sorption properties in Hanford Site soils. Existing literature was searched for the following constituents: tritium, cesium, strontium, cobalt, bismuth, uranium, plutonium, cyanide, and phosphate. The results of this search are presented in Buckmaster (1992).

Laboratory testing was also performed to confirm K_d assumptions made in the vadose zone modeling. The work scope consisted of five batch tests using contaminated groundwater from well 699-50-53A. The contaminants that were tested included: CN

radiolabeled with a C-14 tracer, Co-60, Sr-90/Cs-137, Pu, and Tc-99. Soil samples were collected during the installation of wells 699-52-57 and 699-55-55. Results of these tests are presented in Appendix F.

2.4 ECOLOGICAL INVESTIGATION

The ecological investigation for the first phase of the 200 BP-1 RI consisted of the characterization of potential wildlife receptor organisms in the vicinity of the operable unit. Data collection activities performed under the wildlife ecological investigation included the compilation of existing biological information, and a reconnaissance survey of the operable unit. These two activities are summarized below.

Characterization of human populations was not included as an RI task, however, it was necessary to support performance of the risk assessment. Human population characterization was conducted by compiling existing demographic, land use and cultural information. This information was obtained from government sources, as well as from existing Hanford Site documents and reports. An interpretation of the ecology of the operable unit, from both a human and wildlife perspective, is presented in Section 3.7 (Ecological Characteristics).

2.4.1 Biological Data Compilation

Existing biological information was compiled to develop a general understanding of the wildlife ecology of the operable unit and vicinity. A description of the general wildlife ecology of the Hanford Site was obtained from the Hanford Site National Environmental Policy Act (NEPA) Characterization (Cushing 1990), and a recent annual environmental monitoring report for the Hanford Site (Jaquish and Bryce 1990). Operable-unit-specific information pertaining to the occurrence of birds and other organisms was found in various reports (Brandt 1988, Eberhart et al. 1982, Fitzner et al. 1981, Gano and States 1982, Rogers and Rickard 1977, and Sackschewsky et al. 1992).

An endangered and threatened species survey for the 200 BP-1 operable unit was conducted. The purpose of this survey was to determine the potential for adverse impacts to protected species, or habitats critical to their existence, from either operable unit contaminant releases or subsequent remediation efforts. This survey consisted of a review of literature published by the Washington State Department of Wildlife (DOW) and the Washington State Department of Natural Resources (DNR). The DOW and DNR were requested to review their records for sightings of endangered or threatened animal or plant species in the vicinity of the operable unit. The results of these record reviews are contained within letters received from DOW and DNR (see Appendix G).

2.4.2 Biological Survey

Westinghouse Hanford Company conducted a reconnaissance biotic survey of the 200 BP-1 operable unit in 1989. The survey was conducted to locate and evaluate any evidence of, or potential for, uptake of toxic substances by plants or animals. Hanford Site biologists documented evidence of impacted plants by location and species. Observations

were also made of evidence of small mammal and bird occurrences and animal-burrowing activities. The results of this survey are provided in a project memorandum (Appendix G).

2.5 DATA VALIDATION AND QUALITY

Sampling procedures, analytical methods, and data validation requirements for environmental media were specified in the Quality Assurance Project Plan (QAPJP) of the 200-BP-1 Phase 1 RI/FS Work Plan (DOE-RL 1990). Data validation was conducted in accordance with established procedures and guidelines developed for Chemical Analyses (WHC 1992a, and WHC 1992b). This section describes the status of the data validation and describes which data has been validated to date.

The RI was performed using data contained within the 200-BP-1 project database as of February 28, 1992. These data include both validated and unvalidated data. Of the data included, over 99% of Task 7 groundwater analytical data and Tasks 2, 3, 4 and 6 soil analytical data had been validated. No Task 10 or 12 data had been validated.

As data from each data package was validated, a validation summary report was completed and attached to the data package. These data packages with attached validation summary reports were then delivered to WHC for filing at the EDMC where they can be obtained for review.

Appendix J provides a summary of the data quality evaluation. This evaluation includes a summary of the samples validated, field QC samples, analyses performed and results, and the number of datapoints that were deemed valid. The evaluation addresses the work plan requirements for precision, accuracy, representativeness, comparability and completeness (PARCC).

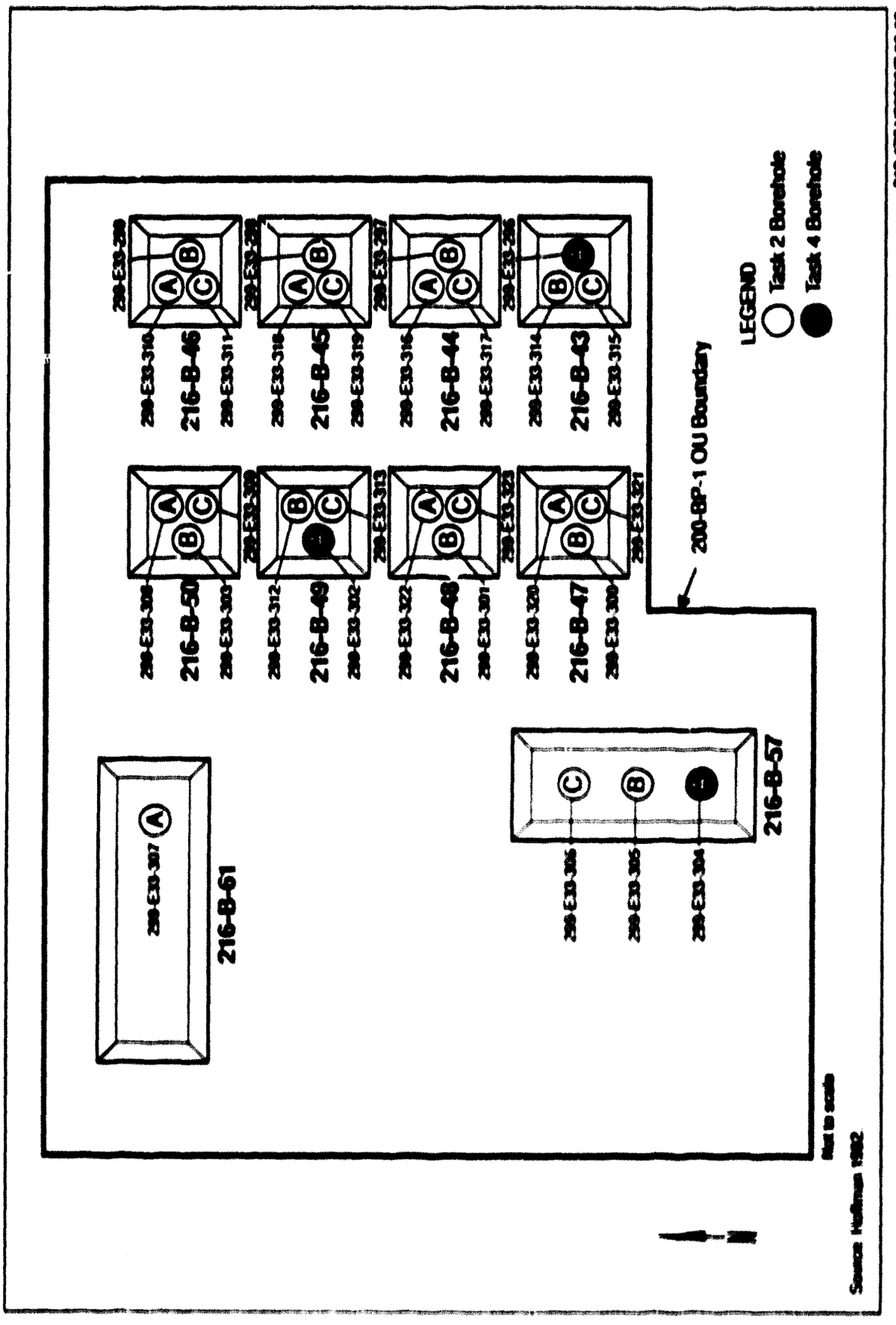


Figure 2-1. Task 2 and 4 Boring Locations

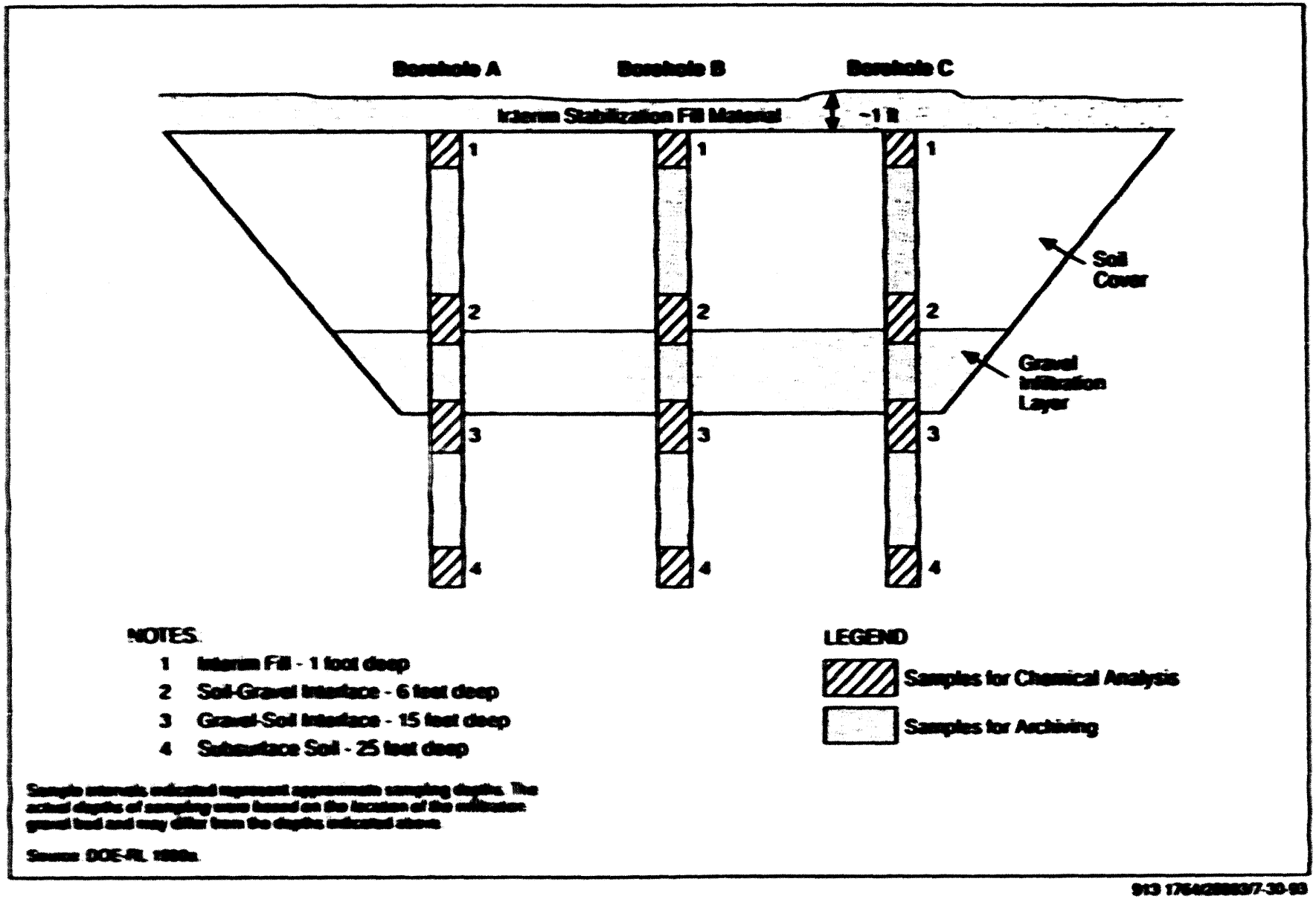
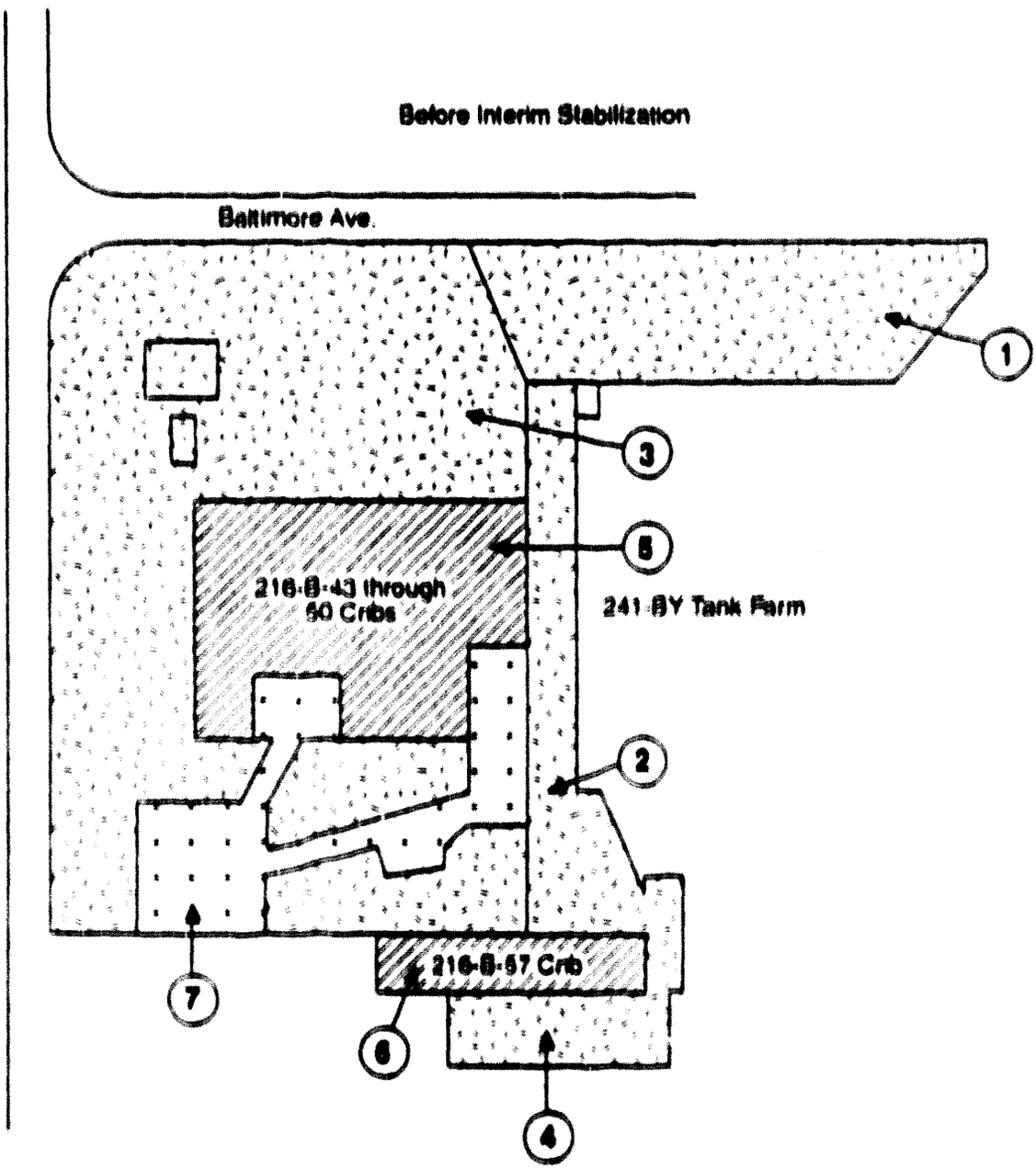


Figure 2-2. Schematic Diagram for Task 2 — Source Sampling and Analysis.



Source: Hayward 1998

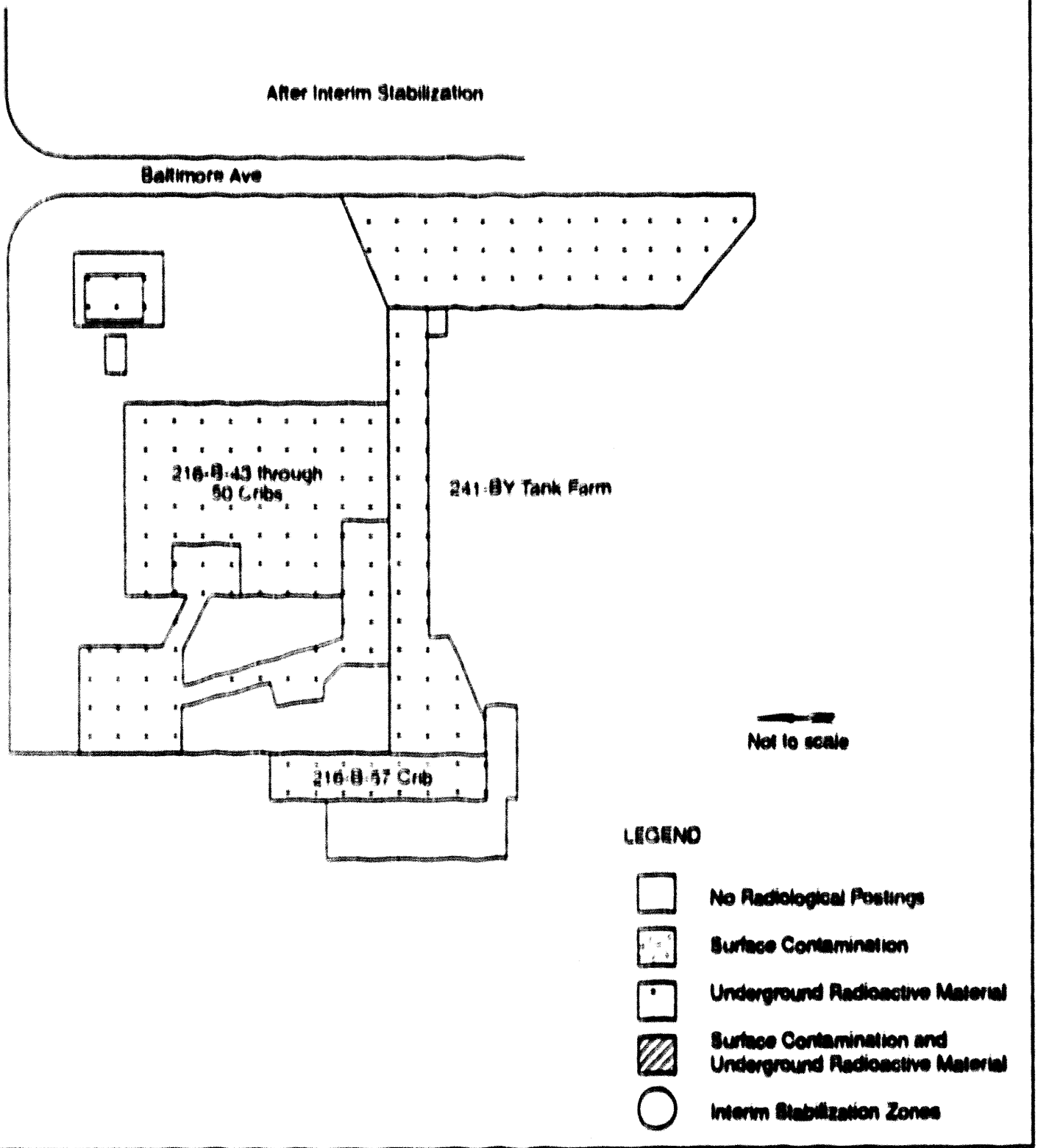
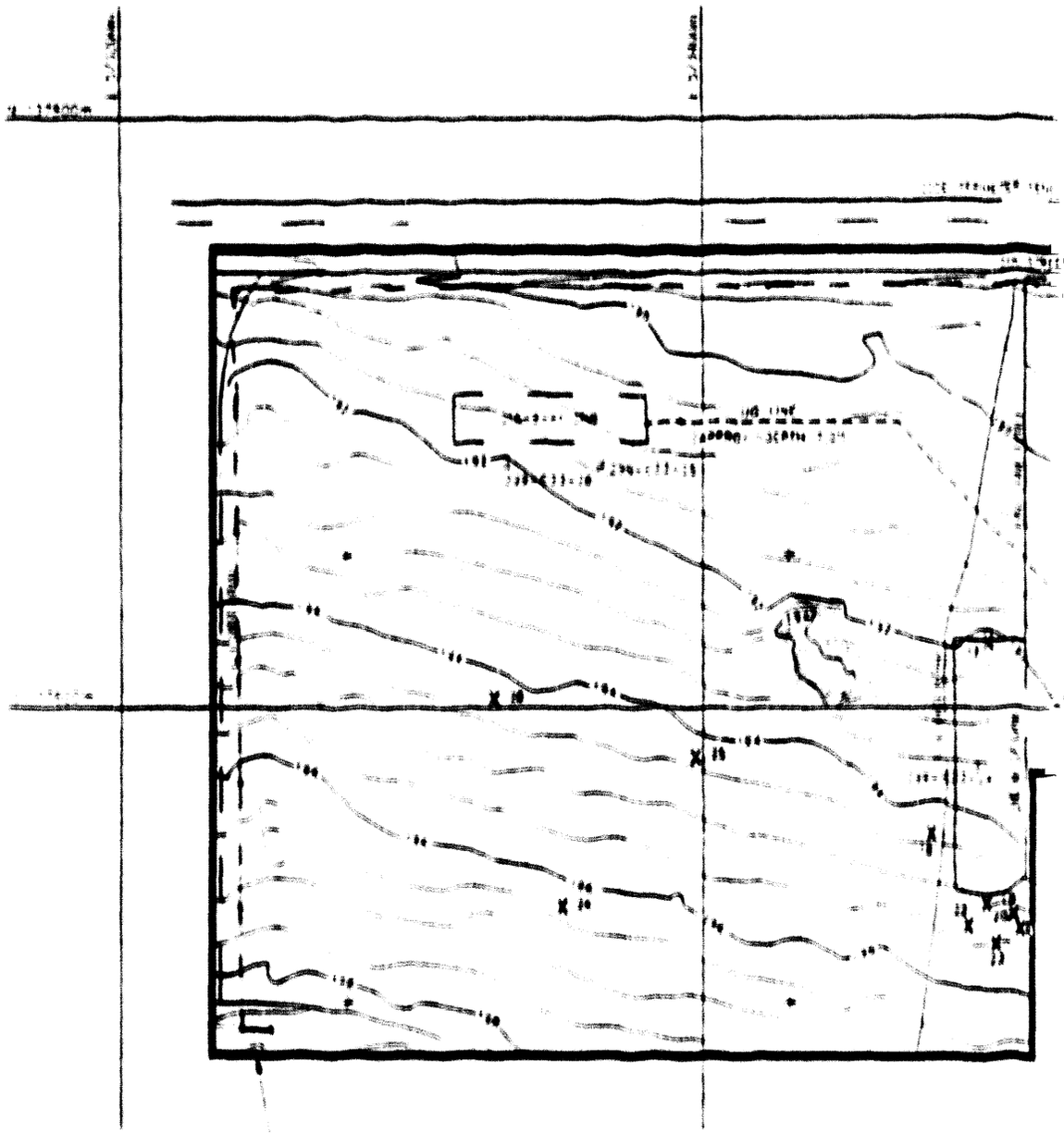
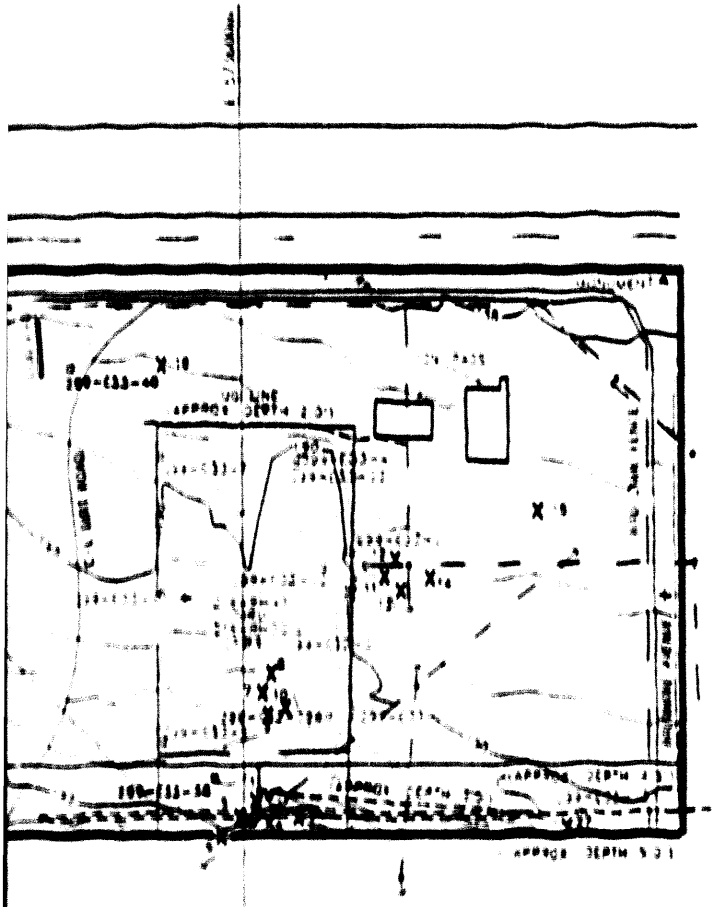


Figure 2-3. Surface Contamination Before and After Interim Stabilization within the 200-BP-1 Operable Unit.



200-[illegible] OPERABLE UNIT BOUNDARY



Legend:

- Manhole
- 299-E33-23 Ground Water Monitoring Wells
- Survey Control Points
- Hanford Plant Coordinate System Grid Ticks
- Fence Lines
- Surface Soil Sampling Locations
- Power Lines
- Underground Pipelines
- Guy Anchors

Contours in Meters AMSL (NGVD 29)
1m = 3.28ft

Grid in Lambert coordinates (meters), NAD83.

N

0 60 120 METERS
0 200 400 FEET

\CAD\8131764\031891-A 8-17-93 9:24

Figure 2-4. Topography, Surface Soil Sampling Locations, and Utilities of the 200-BP-1 Operable Unit.

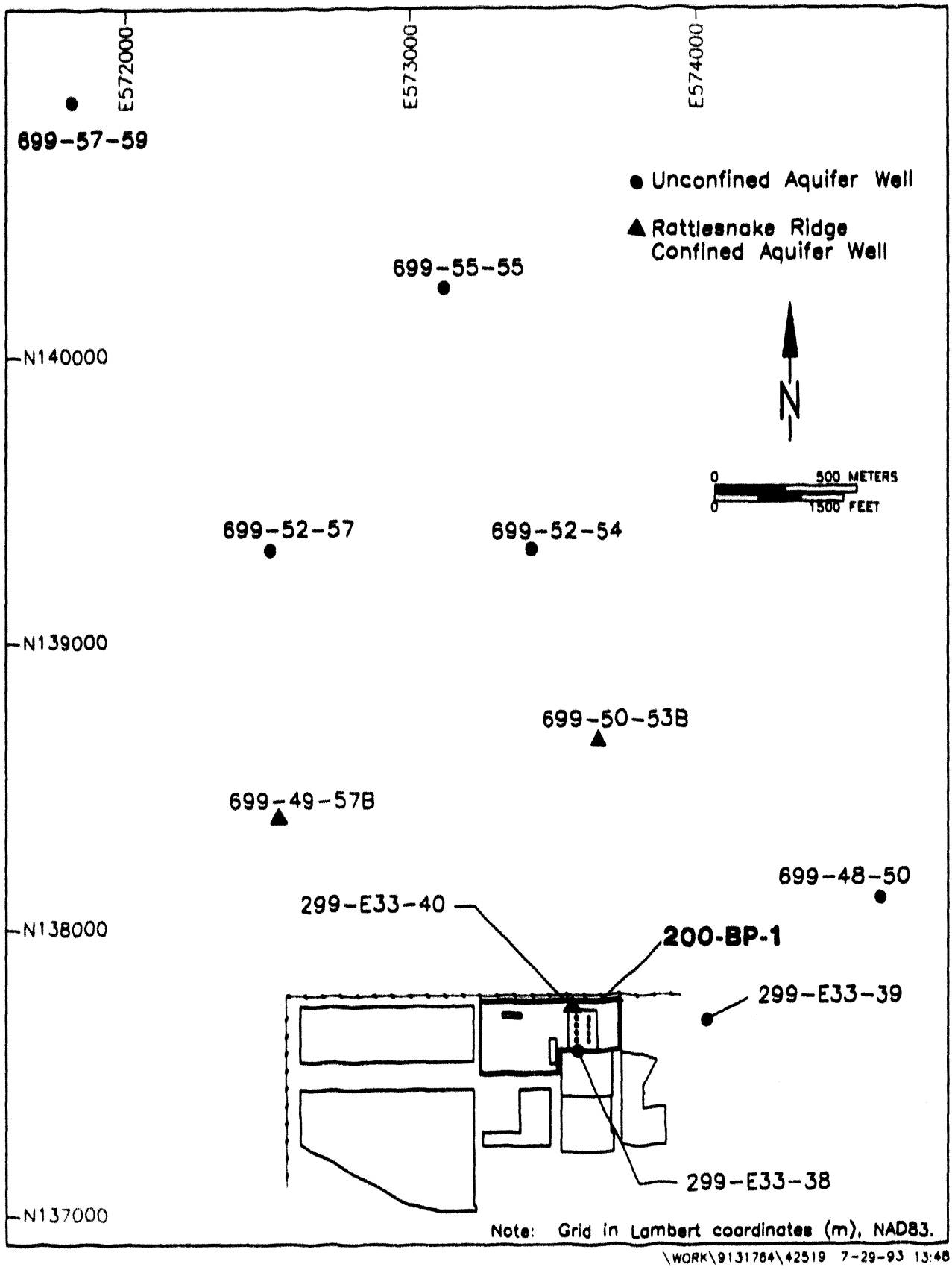


Figure 2-5. Monitoring Well Locations for Task 6.

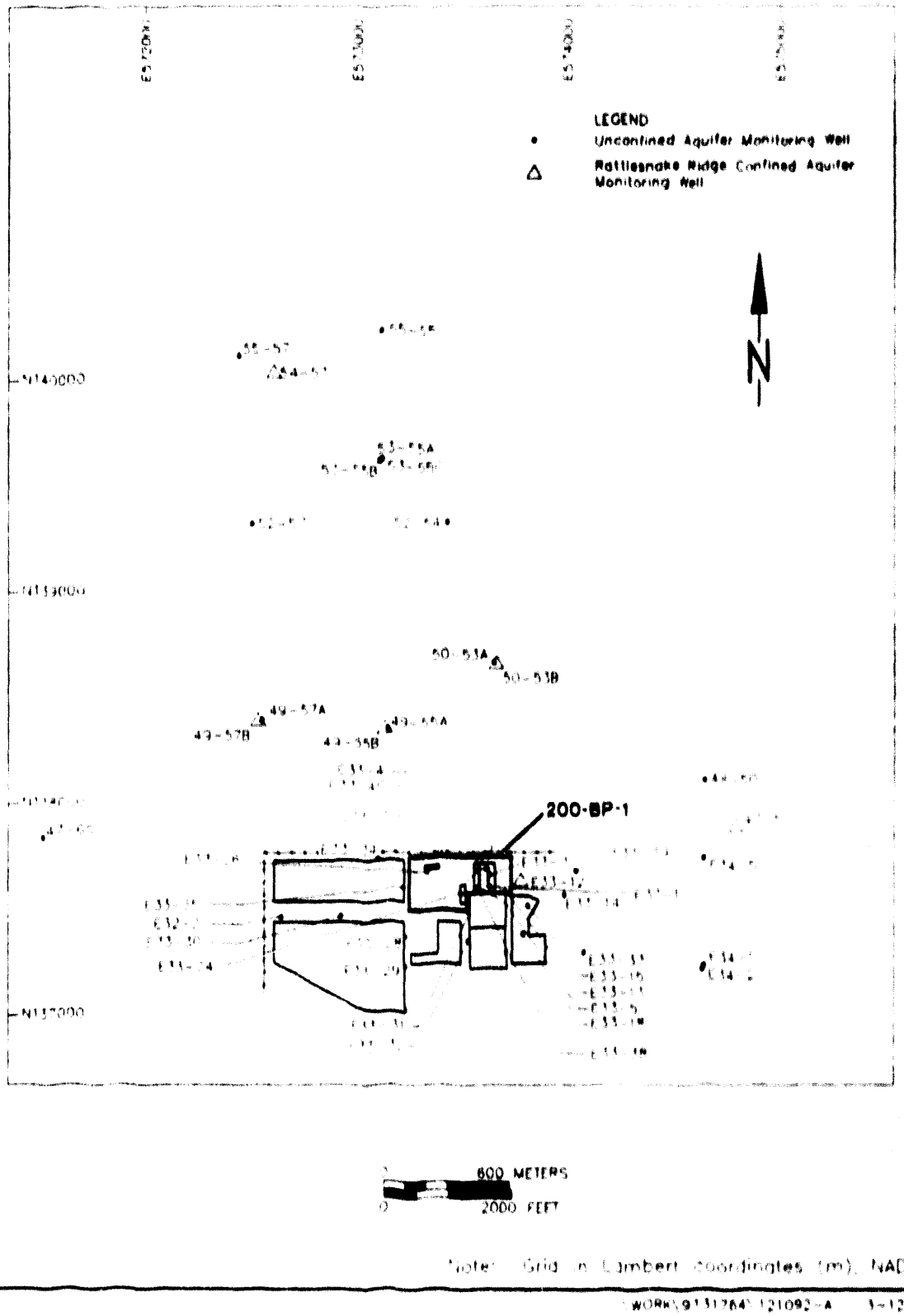


Figure 2-6. Task 7 Groundwater Monitoring Well Locations.

Table 2-1. Summary of Remedial Investigation Phase I Data Collection Activities[e]. (Sheet 1 of 2)

Task No. and description	#of Bore-holes/ depth	Wells	Samples[a]	Analyses	Radiological survey	Boreholes geophysics
Task 2--Source sampling and analysis	26/10 m 2/15 m	N/A[b]	109 (40) 113 (32) 109 (32) 129 (33) 114 (31) 115 (31) 116 (33) 79 (25)	TCL - Volatile Organic Analyses TCL - Semivolatiles TCL - Pesticides Anions TAL Metals and Bismuth Radioisotopes PNL-CN Weston-CN	N/A	X
Task 3--Surface soil sampling and analysis	N/A	N/A	26 (2) 41 (4) 26 (2) 16 (2) 41 (4)	Anions Bismuth and Selenium Radioisotopes PNL-CN Weston-CN	X	N/A
Task 4--Vadose zone soil sampling and analysis	3/~70 m	N/A	24 (4) 24 (4) 24 (4) 33 (4) 24 (4) 24 (4) 22 (3) 23 (3)	TCL - Volatile Organic Analyses TCL - Semivolatiles TCL - Pesticides Anions TAL Metals and Bismuth Radioisotopes PNL-CN Weston-CN	N/A	X
Task 5--Seismic refraction survey	N/A	N/A	N/A	N/A	N/A	N/A
Task 6--Installation of monitoring wells	N/A	10	32(2) 17(16) 17(16)	TOC anions with bismuth, selenium, total and free cyanide radioisotopes.	N/A	X
Task 7--Groundwater sampling and analysis[d]	N/A	44	36-44 (quarterly for 5 qrts)	TCL - Volatiles TCL - Semivolatiles TCL - Pesticides/PCBs TAL Metals plus Bismuth, Silicon, Tin and Strontium Wet Chemistry[e] Radioisotopes PNL-CN Weston-CN	N/A	N/A

Table 2-1. Summary of Remedial Investigation Phase I Activities. (Sheet 2 of 2)

Task No. and description	#of Bore-holes/ depth	Wells	Samples[a]	Analyses	Radiological survey	Boreholes geophysics
Task 8--Site topographic map	N/A	N/A	N/A	N/A	N/A	N/A
Task 9--Biota evaluation	N/A	N/A	N/A	N/A	N/A	N/A
Task 10--Column leach tests	N/A	N/A	N/E[f]	N/E	N/A	N/A
Task 11--Aquifer tests	N/A	4	N/A	N/A	N/A	N/A
Task 12--Sorption tests	N/A	N/A	N/E	N/E	N/A	N/A

[a]Quality control samples in parentheses.
[b]N/A = Not applicable.
[c]TCL = Target compound list.
[d]See Table 2-5 for a detailed summary of wells sampled and analyses performed each quarter.
[e]Numbers of samples indicated herein represent sample results provided to Golder Associates as of 12-1-92. This table does no necessarily list all samples collected or analyzed.
[f]N/E = not evaluated
[g]Wet chemistry includes major anions, alkalinity, hardness, pH, specific conductence, total dissolved solids and total organic carbon.

Table 2-2. Summary Table of 200-BP-1 Tasks 2 and 4 Drilling. (Sheet 1 of 2)

Crib No.	Official Borehole No.	Start date	End drilling date	End abandonment date	Total depth m(ft)	Lambert coordinates (m) NAD'83	Elevation NGVD'29 m(ft)	Top of crib m(ft)	Bottom of crib m(ft)	No. of chemical samples	No. of physical samples	No. of archive samples
216-B-43 A	299-E33-296	11/7/91	1/23/92	3/30/92	69.2 (227.0)	N: 137,613.85 E: 573,628.12	191.7 (628.9)	3.5 (11.6)	5.3 (17.5)	12	31	63
216-B-43 B	299-E33-314	4/23/92	4/24/92	5/15/92	9.3 (30.5)	N: 137,639.88 E: 573,627.67	191.8 (629.2)	3.5 (11.5)	5.5 (18.0)	4	–	8
216-B-43 C	299-E33-315	4/29/92	4/30/92	5/19/92	9.6 (31.5)	N: 137,611.59 E: 573,625.89	191.7 (629.0)	3.5 (11.5)	5.8 (19.0)	4	–	8
216-B-44 A	299-E33-316	3/18/92	3/19/92	5/6/92	9.6 (31.5)	N: 137,642.45 E: 573,625.42	191.8 (629.2)	3.4 (11.0)	6.1 (20.0)	4	–	8
216-B-44 B	299-E33-297	3/25/92	3/30/92	5/13/92	10.1 (33.0)	N: 137,639.88 E: 573,627.67	191.8 (629.4)	3.2 (10.5)	6.0 (19.8)	4	–	6
216-B-44 C	299-E33-317	4/3/92	4/8/92	5/14/92	9.6 (31.5)	N: 137,637.78 E: 573,625.01	191.8 (629.3)	3.4 (11.0)	6.1 (20.0)	4	–	4
216-B-45 A	299-E33-318	1/20/92	1/27/92	4/8/92	9.1 (30.0)	N: 137,668.29 E: 573,624.91	191.4 (627.8)	3.4 (11.2)	5.2 (17.0)	4	–	7
216-B-45 B	299-E33-298	2/28/92	3/5/92	4/14/92	9.0 (29.5)	N: 137,665.38 E: 573,627.17	191.3 (627.7)	3.4 (11.0)	5.9 (19.5)	5	–	9
216-B-45 C	299-E33-319	3/10/92	3/11/92	4/15/92	9.0 (29.5)	N: 137,663.38 E: 573,624.83	191.5 (628.2)	3.4 (11.0)	5.8 (19.0)	4	–	3
216-B-46 A	299-E33-310	11/21/91	12/5/91	12/4/91	9.8 (32.0)	N: 137,693.95 E: 573,625.39	190.9 (626.4)	3.4 (11.0)	5.0 (16.5)	4	–	8
216-B-46 B	299-E33-299	12/10/91	12/17/91	2/28/92	10.7 (35.0)	N: 137,691.91 E: 573,627.58	191.0 (626.7)	3.2 (10.5)	5.2 (17.0)	4	–	11
216-B-46 C	299-E33-311	1/7/92	1/10/92	3/2/92	9.0 (29.5)	N: 137,689.45 E: 573,625.30	191.1 (627.0)	3.4 (11.0)	5.2 (17.0)	4	–	9
216-B-47 A	299-E33-320	4/14/92	4/22/92	5/22/92	10.7 (35.0)	N: 137,615.88 E: 573,582.13	192.6 (631.8)	4.3 (14.0)	6.9 (22.5)	4	–	5
216-B-47 B	299-E33-300	4/27/92	5/4/92	5/28/92	11.4 (37.5)	N: 137,613.86 E: 573,579.72	192.6 (631.8)	4.7 (15.5)	6.8 (22.2)	4	–	8
216-B-47 C	299-E33-321	5/16/92	5/12/92	5/29/92	10.2 (33.5)	N: 137,611.26 E: 573,581.57	192.7 (632.1)	4.5 (14.8)	6.7 (22.0)	4	–	4

Table 2-2. Summary Table of 200-BP-1 Tasks 2 and 4 Drilling. (Sheet 2 of 2)

Crib No.	Official Borehole No.	Start date	End drilling date	End abandonment date	Total depth m(ft)	Lambert coordinates (m) NAD'83	Elevation NGVD'29 m(ft)	Top of crib m(ft)	Bottom of crib m(ft)	No. of chemical samples	No. of physical samples	No. of archive samples
216-B-48 A	299-E33-322	3/11/92	3/16/92	4/28/92	9.3 (30.5)	N: 137,642.09 E: 573,581.72	191.8 (629.4)	3.5 (11.5)	5.9 (19.5)	4	–	3
216-B-48 B	299-E33-301	3/20/92	3/25/92	4/30/92	10.7 (35.0)	N: 137,639.69 E: 573,579.47	191.9 (629.6)	3.8 (12.5)	6.1 (20.0)	4	–	6
216-B-48 C	299-E33-323	3/31/92	4/2/92	5/4/92	9.6 (31.5)	N: 137,637.50 E: 573,581.90	192.0 (629.9)	3.8 (12.5)	5.9 (19.5)	4	–	5
216-B-49 A	299-E33-302	7/25/91	11/5/92	1/10/92	68.0 (223.0)	N: 137,665.73 E: 573,579.94	190.6 (625.3)	4.1 (13.5)	5.2 (17.0)	11	27	44
216-B-49 B	299-E33-312	11/8/91	11/20/91	12/26/91	9.8 (32.0)	N: 137,667.83 E: 573,582.36	190.6 (625.3)	3.7 (12.0)	5.5 (18.0)	4	–	10
216-B-49 C	299-E33-313	1/23/92	3/2/92	4/2/92	9.0 (29.5)	N: 137,663.37 E: 573,581.87	190.5 (625.5)	3.4 (11.0)	5.8 (19.0)	4	–	6
216-B-50 A	299-E33-308	11/4/91	11/12/91	12/10/91	8.8 (29.0)	N: 137,693.80 E: 573,582.37	190.7 (625.0)	3.5 (11.6)	5.0 (16.5)	4	–	11
216-B-50 B	299-E33-303	10/2/91	10/9/91	10/11/91	10.1 (33.0)	N: 137,691.63 E: 573,579.92	190.5 (624.9)	4.3 (14.0)	5.6 (18.5)	4	–	9
216-B-50 C	299-E33-309	10/17/91	10/23/91	10/31/91	10.1 (33.0)	N: 137,689.16 E: 573,582.08	190.6 (625.4)	3.8 (12.5)	5.9 (19.5)	4	–	10
216-B-57 A	299-E33-304	6/25/91	8/27/91	10/25/91	71.0 (233.0)	N: 137,563.32 E: 573,496.64	194.1 (636.8)	4.0 (13.0)	5.0 (16.5)	12	89	–
216-B-57 B	299-E33-305	9/13/91	9/19/91	9/23/91	15.2 (50.0)	N: 137,584.72 E: 573,500.05	193.9 (636.0)	3.3 (10.8)	5.5 (18.0)	6	–	18
216-B-57 C	299-E33-306	8/30/91	9/6/91	9/10/91	15.2 (50.0)	N: 137,605.89 E: 573,496.80	193.4 (634.4)	3.4 (11.0)	5.7 (18.8)	5	–	21
216-B-61 A	299-E33-307	5/9/91	5/20/91	5/21/91	9.2 (30.2)	N: 137,698.77 E: 573,496.80	190.7 (625.7)	2.9 (9.5)	4.6 (15.0)	4	9	–

NAD'83 = North American Datum 1983
NGVD'29 = National Geodetic Vertical Datum 1929
- - Means no Samples were obtained
Source: Hoffman 1992

Table 2-3. Summary Table of 200-BP-1 Task 6 Drilling and Well Installation.

Well No.	Start date	End drilling date	Date of completion	Total depth m(ft)	Depth to water m(ft)	Aquifer screened	Screen interval m(ft)	Lambert coordinates (m) NAD83	Casing elev. NGVD 29 m(ft)	No. of chemical samples	No. of physical samples	No. of archive samples
299-E33-38	11/16/90	1/8/91	2/1/91	73.8 (239.6)	69.0 (226.5)	Unconfined	66.6-73.8 (218.5-239.6)	N: 137,594.78 E: 573,591.39	192.6 (631.9)	1	13	43
299-E33-39	12/3/90	1/18/91	2/8/91	70.1 (230.1)	66.7 (218.7)	Unconfined	63.5-69.9 (208.2-229.2)	N: 137,637.65 E: 573,843.75	190.0 (623.3)	–	8	45
299-E33-40	11/18/90	2/11/91	4/1/91	96.9 (318.0)	66.3 (217.6)	Confined	89.6-92.9 (293.9-304.9)	N: 137,723.38 E: 573,546.44	190.3 (624.3)	2	10	50
699-48-50	4/13/90	5/8/90	9/27/90	60.0 (197.0)	50.3 (165.0)	Unconfined	48.6-54.8 (159.4-179.7)	N: 138,227.38 E: 574,817.75	175.0 (574.1)	–	5	59
699-49-57B	4/15/90	10/9/90	11/9/90	70.2 (230.3)	45.4 (149.0)	Confined	66.8-70.0 (219.3-229.7)	N: 138,381.34 E: 572,536.62	169.5 (556.0)	6	10	12
699-50-53B	4/5/90	10/15/90	10/12/91	68.6 (225.0)	46.2 (151.7)	Confined	65.4-68.5 (214.7-224.7)	N: 138,659.81 E: 573,655.63	170.0 (557.6)	4	11	28
699-52-54	4/25/90	5/22/90	11/26/91	51.0 (167.4)	49.5 (162.4)	Unconfined	47.7-50.9 (156.5-166.9)	N: 139,193.59 E: 573,254.43	173.3 (568.4)	–	10	22
699-52-57	9/28/90	10/26/90	11/22/91	50.4 (165.5)	47.5 (155.8)	Unconfined	45.4-48.6 (149.1-159.5) 42.4-48.6* (139.2-159.4)	N: 139,115.59 E: 572,761.58	171.2 (561.8)	7	28	12
699-55-55	10/1/90	12/19/90	12/3/91	95.1 (312.0)	47.9 (157.3)	Unconfined	45.2-57.6 (148.4-189.3) 46.0-52.2* (150.8-171.1)	N: 140,149.99 E: 573,227.11	171.8 (563.6)	10	41	18
699-57-59	3/17/92	4/8/92	4/16/92	57.6 (189.0)	52.2 (171.1)	Unconfined	50.6-56.8 (166.0-186.3)	N: 140,504.00 E: 571,830.36	175.6 (576.3)	–	6	38

*8 in. telescoping screen.
NAD83 = North American Datum 1983
NGVD29 = National Geodetic Vertical Datum 1929
Source: Hoffman 1992
– means no samples obtained

Table 2-4. 200-BP-1 Spectral Gamma-Ray Log Surveys (Tasks 2, 4 and 6).

Permanent Hole ID	Temporary Hole ID	Final Date	Survey Depth meters (ft)	Man-Made Radionuclides			
				Cs137	Co60	Sb125	Eu154
299-E33-296	216-B-43A	Jan 27, 92	67.7 (222)	Cs137	Co60		
299-E33-297	216-B-44B	Apr 21, 92	9.0 (29.5)	Cs137	Co60		Eu154
299-E33-298	216-B-45B	Mar 18, 92	7.6 (25)	Cs137			
299-E33-310	216-B-46A	Jan 20, 92	8.8 (29)	CS137	Co60	Sb125	
299-E33-299	216-B-46B	Jan 20, 92	9.4 (31)	Cs137	Co60	Sb125	
299-E33-311	216-B-46C	Jan 21, 92	7.9 (26)	Cs137	Co60	Sb125	
299-E33-300	216-B-47B	May 19, 92	10.5 (34.5)	Cs137			
299-E33-301	216-B-48B	Apr 14, 92	9.8 (32)	Cs137			
299-E33-302	216-B-49A	Nov 26, 91	67.1 (220)	Cs137	Co60	Sb125	
299-E33-312	216-B-49B	Dec 10, 91	8.5 (28)	Cs137	Co60	Sb125	
299-E33-308	216-B-50A	Dec 12, 91	7.3 (24)	Cs137	Co60		
299-E33-303	216-B-50B	Oct 10, 91	8.8 (29)	Cs137	Co60	Sb125	
299-E33-309	216-B-50C	Oct 25, 91	9.1 (30)	Cs137	Co60	Sb125	
299-E33-304	216-B-57A	Sep 15, 91	70.1 (230)	Cs137			
299-E33-305	216-B-57B	Sep 20, 91	14.3 (47)	Cs137			
299-E33-306	216-B-57C	Sep 9, 91	15.1 (49.5)	.	.	.	None
299-E33-307	216-B-61A	May 20, 91	8.4 (27.5)	Cs137			
299-E33-1		Sep 25, 91	71.3 (234)	Cs137	Co60		
299-E33-2		Sep 30, 91	60.8 (199.5)	Cs137	Co60		Eu154
299-E33-3		Sep 27, 91	70.7 (232)	Cs137	Co60	Sb125	Eu154
299-E33-7		Sep 13, 91	69.5 (228)	Cs137	Co60	Sb125	
299-E33-13		Jul 22, 92	70.6 (231.7)	Cs137	Co60		
299-E33-22		Oct 7, 91	56.4 (185)	Cs137	Co60		
299-E33-23		Oct 3, 91	67.8 (222.5)	Cs137	Co60	Sb125	
299-E33-24		Jul 15, 92	75.2 (246.7)	Cs137	Co60		
299-E33-38		Jan 9, 91	71.0 (233)	Cs137	Co60	Sb125	
299-E33-40		Feb 13, 91	96.2 (315.5)	Cs137	Co60		

Source: Price (1992)
Blank means isotope was not analyzed in logging effort.

Table 2-8. Task 7 Groundwater Monitoring Wells. (Sheet 1 of 7)

WELL NUMBER	QTR*	AQUIFER	METALS	VOA	SVOA	PEST/PCB	WCHEM	PNL-CN	RAD	WEB-CN
299-E33-08	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V	V	V	V
	4	U	V				V		V	V
	5	U	V				V		V	V
299-E33-01	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V		V	V
299-E33-08	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V		V	V
	5	U	V				V		V	V
299-E33-04	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V		V	V
	4	U	V				V		V	V
	5	U	V				V		V	V
299-E33-08	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V	V	V	V
	4	U	V				V	V	V	V
	5	U	V				V		V	V
299-E33-07	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V	V	V	V
	5	U	V				V		V	V
299-E33-12	1	C	V	V	V	V	V	V	V	V
	2	C	V				V	V	V	V
	3	C	V				V		V	V
	4	C	V				V	V	V	V
	5	C	V				V		V	V
299-E33-13	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V

Table 2-5. Task 7 Groundwater Monitoring Wells. (Sheet 2 of 7)

WELL NUMBER	QTR*	AQUIFER	METALS	VOA	SVOA	PEST/PCB	WCHEM	PNL-CN	RAD	WBS-CN
	3	U	V				V		V	V
	4	U	V				V		V	V
	5	U	V				V		V	V
299-E33-14	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V		V	V
	4	U	V				V		V	V
	5	U	V				V		V	V
299-E33-15	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	N	V
	3	U	V				V		V	V
	4	U	V				V		V	V
	5	U	V				V		V	V
299-E33-18	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	N	V
	3	U	V				V		V	V
	4	U	V				V		V	V
	5	U	V				V		V	V
299-E33-24	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V		V	V
	4	U	V				V		V	V
	5	U	V				V		V	V
299-E33-26	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V		V	V
	4	U	V				V		V	V
	5	U	V				V		V	V
299-E33-28	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V	V	V	V
	4	U	V				V		V	V

Table 2-8. Task 7 Groundwater Monitoring Wells. (Sheet 3 of 7)

WELL NUMBER	QTR*	AQUIFER	METALS	VOA	SVOA	PEST/PCB	WCHEM	PNL-CN	RAD	WBS-CN
	2	U	V				V		V	V
299-E33-29	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V	V	V	V
	4	U	V				V		V	V
	5	U	V				V		V	V
299-E33-30	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V	V	V	V
	4	U	V				V		V	V
	5	U	V				V		V	V
299-E33-31	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V		V	V
	4	U	V				V		V	V
	5	U	V						V	V
299-E33-32	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V		V	V
	4	U	V				V		V	V
	5	U	V				V		V	V
299-E33-33	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V		V	V
	4	U	V				V		V	V
	5	U	V				V		V	V
299-E33-34	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V		V	V
	4	U	V				V		V	V
	5	U	V				V		V	V
299-E33-35	1	U	V	V	V	V	V	V	V	V

Table 2-5. Task 7 Groundwater Monitoring Wells. (Sheet 4 of 7)

WELL NUMBER	QTR	AQUIFER	METALS	VOA	SVOA	PEST/PCB	WATER	PNL-CN	RAD	WEK-CN
	2	U	V				V	V	V	V
	3	U	V				V		V	V
	4	U	V				V		V	V
	5	U	V				V		V	V
[illegible]	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V		V	V
	4	U	V				V		V	V
	5	U	V				V		V	V
[illegible]	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V		V	V
	4	U	V				V		V	V
	5	U	V				V		V	V
[illegible]	1	C	V	V	V	V	V	V	V	V
	2	C	V				V	V	V	V
	3	C	V				V		V	V
	4	C	V				V		V	V
	5	C	V				V		V	V
[illegible]	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V		V	V
	4	U	V				V		V	V
	5	U	V				V		V	V
[illegible]	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V		V	V
	4	U	V				V		V	V
	5	U	V				V		V	V
[illegible]	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V		V	V

Table 2-8. Task 7 Groundwater Monitoring Wells. (Sheet 5 of 7)

WELL NUMBER	QTR*	AQUIFER	METALS	VOA	SVOA	PEST/ PCB	WQ/HEM	PNL/CN	RAD	WEK/CN
	4	U	V				V		V	V
	5	U	V				V		V	V
699-47-59	1	C	V	V	V	V	V	V	V	V
	2	C	V				V	V	V	V
	3	C	V				V			
	5	C	V				V		V	V
699-47-60	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V		V	V
	4	U	V				V		V	V
	5	U	V				V		V	V
699-48-59	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V			
	4	U	V				V		V	V
	5	U	V				V		V	V
699-49-58A	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V	V	N	V
	4	U	V				V	V	V	V
	5	U	V				V	V	V	V
699-49-58B	1	C	V	V	V	V	V	V	V	
	2	C	V				V	V	V	V
	4	C	V				V		V	V
	5	C	V				V		V	V
699-49-57A	1	U	V	V	V	V	V	V	V	
	2	U	V				V	V	V	V
	3	U	V				V	V	N	V
	5	U	V				V	V	V	V
699-49-57B	1	C	V	V	V	V	V	V	V	V
	2	C	V				V	V	V	V
	3	C	V				V		N	V

Table 2-5. Task 7 Groundwater Monitoring Wells. (Sheet 6 of 7)

WELL NUMBER	QTR*	AQUIFER	METALS	VOL.	SVOL.	PEST/ PCB	WCHEM	PNL-CN	RAD	WBS-CN
	4	C	V				V		V	V
	5	C	V				V		V	V
699-50-53A	1	U	V	V	V	V	V	V	V	
	2	U	V				V	V	V	V
	3	U	V				V	V	V	V
	4	U	V				V	V	V	V
	5	U	V				V		V	V
699-50-53B	1	C	V	V	V	V	V	V	V	
	2	C	V				V	V	V	V
	3	C	V				V		N	V
	4	C	V				V		V	V
	5	C	V				V		V	V
699-52-54	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V	V	V	V
	5	U	V				V		V	V
699-52-57	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V		N	V
	5	U	V				V		V	V
699-53-55A	1	U	V	V	V	V	V	V	V	
	2	U	V				V	V	V	V
	3	U	V				V		V	V
	4	U	V				V		V	V
	5	U	V				V		V	V
699-53-55B	1	U	V	V	V	V	V	V	V	
	2	U	V				V	V	V	V
	3	U	V				V		V	V
	4	U	V				V		V	V
	5	U	V				V		V	V
699-53-55C	1	U	V	V	V	V	V	V	V	
	2	U	V				V	V	V	V

Table 2-5. Task 7 Groundwater Monitoring Wells. (Sheet 7 of 7)

WELL NUMBER	QTR*	AQUIFER	METALS	VOL	SVOL	PEST/ PCB	WCHEM	PNL-CN	RAD	WES-CN
	3	U	V				V	V	V	V
	5	U	V				V	V	V	V
699-54-57	1	C	V	V	V	V	V	V	V	
	2	C	V				V	V	V	V
	3	C	V				V		N	V
	5	C	V				V		V	V
699-55-55	1	U	V	V	V	V	V	V	V	V
	2	U	V				V	V	V	V
	3	U	V				V		N	V
	4	U	V				V		V	V
	5	U	V				V		V	V
699-55-57	1	U	V	V	V	V	V	V	V	
	2	U	V				V	V	V	V
	3	U	V				V	V	N	V
	4	U	V				V	V	V	V
	5	U	V				V	V	V	V

Notes:
*1st Quarter Sampling Period is 1/91 - 4/91
2nd Quarter Sampling Period is 5/91 - 7/91
3rd Quarter Sampling Period is 8/91 - 9/91
4th Quarter Sampling Period is 10/91 - 12/91
5th Quarter Sampling Period is 1/92 - 4/92

U = Unconfined Aquifer Well
C = Confined Aquifer Well
V = Validated Data
N = Not Validated Data

PNL-CN means cyanide analyzed by PNL
WES-CN means cyanide analyzed by Weston
VOL means volatile analysis
SVOL means semivolatile analysis
PEST/PCB means pesticide and PCB analysis
WCHEM means wet chemistry analysis
RAD means radionuclide analysis
Blank means sample was not analyzed

3.0 PHYSICAL CHARACTERISTICS

This chapter provides a description of the relevant physical characteristics of the 200-BP-1 operable unit. Descriptions are presented of the waste source characteristics, meteorology, geology, soils, hydrogeology, and ecology.

3.1 SOURCE FACILITIES

3.1.1 Location

The Hanford Site is a 1,450 km^2 (560 mi^2) tract of land located along the Columbia River in southeastern Washington and covers portions of Benton, Grant, Franklin, and Adams Counties (Figure 1-1). The Hanford Site is located approximately 280 km (174 mi) southeast of Seattle and 210 km (130 mi) southwest of Spokane.

The 200-BP-1 operable unit is located in the approximate center of the Hanford Site, along the northern boundary of the 200 East Area. The 200 East Area is located within the separations area that includes both the 200 West and 200 East Areas. A site plan of the 200-BP-1 operable unit is shown in Figure 3-1. Topography of the operable unit is shown in Figure 2-4.

The 200 East Area has been subdivided into operable units (Figure 3-2). The 200-BP-1 operable unit is bordered by operable units 200-BP-4 to the east, 200-BP-7 and 200-BP-3 to the south, and 200-BP-10 to the west. The 600 Area, which includes all areas at the Hanford Site not located in other designated areas (i.e., 100, 200, 300, 400 and 1100 Areas), borders the 200-BP-1 operable unit to the north.

The 200-BP-1 operable unit is located within the southeast quadrant of Section 34 of T13N and R26E and encompasses a total area of approximately 10 ha (25 ac). The majority of the waste management units within the 200-BP-1 operable unit are concentrated in a 1.6 ha (4 ac) region at the eastern end of the 200-BP-1 operable unit (Figure 3-1). The operable unit is bounded within the Washington State plane (NAD 83) north/south coordinates of N137800m and N137500m and east/west coordinates of E573200m and E573800m.

3.1.2 200-BP-1 Waste Management Units

The 200-BP-1 operable unit includes ten inactive cribs (known as the 216-B cribs) and four unplanned releases (UN) (Figure 3-1). Cribs 216-B-43 through -49 received tributyl phosphate (TBP) supernatant waste generated in the 221-U building. The TBP process was used for the recovery of uranium from wastes generated by the bismuth phosphate ($BiPO_4$) process in the B Plant. Waste sent to cribs 216-B-50 and -57 consisted of storage tank condensate from the in-tank solidification units nos. 1 and 2 (ITS nos. 1 and 2), respectively. Both units were located in the 241-BY Tank Farm. In-tank solidification was accomplished by artificial in-tank heating. The tenth crib (216-B-61) was constructed, but there is no historical evidence that it was ever used or received any wastes (DOE-RL

1990a). Waste generating processes will be discussed in detail in Section 4.1.1 (Waste Generating Processes).

Table 3-1 identifies the cribs and UNs contained within 200-BP-1 and summarizes their periods of operation (or dates of occurrence for UNs, if known) and waste sources. Appendix H provides additional information on the cribs and UNs associated with the 200-BP-1 operable unit, including chemical inventories (DOE-RL 1992a).

3.1.2.1 Facility Construction Characteristics. The disposal cribs were designed to receive, disperse and infiltrate liquid waste effluents underground. Waste effluents were diverted to the cribs via the flush tank and underground pipe and were discharged through pipe perforations and sumps to a gravel bed.

Cribs 216-B-43 through 216-B-50 were all of similar design. Design drawings for the cribs suggest that each consists of four 1.2-m (4-ft) diameter by 1.2-m (4-ft) long concrete pipes placed vertically in a 4.6-m (15-ft) deep excavation. The pipes rest on a 1.5-m (5-ft) thick bed of 7.6-cm (3-in.) gravel in a square pattern with centers spaced 4.6 m (15 ft) apart. The tops of the pipes are set 2.1 to 2.4 m (7 to 8 ft) below grade. The bottom of the excavation is 9.1 by 9.1 m (30 by 30 ft) in area. The surface dimensions are approximately 22.9 by 22.9 m (75 by 75 ft). Each culvert is fed by a 20.3-cm (8-in.) steel pipe coming from a main, forming a chevron pattern. Each culvert has a concrete cover. As-built drawings (Plates 2-3 and 2-4 in DOE-RL 1990a) illustrate the construction details in plan and cross-sectional view for cribs 216-B-43 through -50, respectively.

An as-built drawing for crib 216-B-57 (Plate 2-5 in DOE-RL 1990a) indicates that the crib consists of a 30.5-cm (12-in.) corrugated and perforated steel pipe which runs the length of the 61-m (200-ft) long by 4.6-m (15-ft) wide crib (DOE-RL 1990a). The base of the 3-m (10-ft) deep excavation is at an elevation of 189.3 m (621 ft) and is level. The pipe, which slopes to the north, is 1.14 m (3.75 ft) from the bottom of the excavation at the south end and 0.85 m (2.8 ft) above the bottom of the excavation at the north end. The crib bottom is filled with gravel to a depth 1.2 m (4 ft) above the crib bottom. Side slopes are 1.5:1. The location of underground pipelines are shown on Figure 2-4 (additional details are available on WHC Drawing H-2-78769, Rev. 0).

Additional information regarding construction dimensions of the cribs, primarily with respect to depth, was obtained in the Task 2 and 4 drilling activities. Table 2-2 summarizes the crib top and bottom depths observed in each crib borehole. The tops of the cribs (top of infiltration gravel) were generally encountered at depths of from about 3 to 4.27 m (10 to 14 ft) below land surface. The gravels were between 1.1 to 2.8m (3.5 to 9.3 ft) thick, but averaged about 2.1m (6.8 ft) thick. The crib depths observed in the drilling, therefore, are generally from about 0.6 to 1.83 m (2 to 6 ft) deeper than suggested in the drawings. This may be due, at least partly, to the interim stabilization activities which were performed for the UN-200-E-89 unplanned release. These activities involved placement of clean fill over the cribs, thereby raising the level of the ground surface. Crib 216-B-57, however, appears to have been constructed from 6.5 to almost 9 ft deeper than indicated in the as-built drawing for the crib.

3.1.2.2 Operational Characteristics. Limited amounts of liquid wastes were discharged to the 216-B-43 to -49 cribs because of the appreciable concentrations of radionuclides contained in these waste streams. The discharged effluents were allowed to infiltrate into the underlying native soils. The amount of TBP supernatant waste discharged to any one

of the 216-B-43 through 216-B-49 cribs was limited by the resulting concentration of any "critical nuclide" in the underlying groundwater (Thomas et al. 1956). A critical nuclide was defined as a radioactive isotope with a half-life greater than 3 years. Discharge to a crib was permitted until the concentration of such a nuclide in a groundwater sample collected from an adjacent well, was found to exceed, one-tenth the nuclide's maximum permissible drinking water concentration. For example, the maximum permissible drinking water concentration for cobalt-60 during this period was 400,000 pCi/L. The corresponding maximum groundwater concentration was therefore 40,000 pCi/L. However, cobalt-60 was found exceeding this discharge limit in a monitoring well installed adjacent to the cribs by a factor of more than 100 in early 1956 (Thomas et al. 1956), thereby indicating that the operational plan to control crib discharges did not function as intended in limiting impacts to the underlying groundwater. The "time lag" between disposal of the TBP supernatant waste and the appearance of the radionuclides in the groundwater was not accounted for.

Cribs 216-B-50 and -57 received ITS condensates that were low-level wastes. These ITS waste streams were considered suitable for crib disposal and infiltration in large volumes.

3.1.2.3 Unplanned Releases. Four unplanned releases have been identified (Figure 3-1) within the 200-BP-1 operable unit. These unplanned releases have been designated (DOE-RL 1992a) as the following waste units:

- UN-200-E-9
- UN-200-E-63
- UN-200-E-89
- UN-200-E-110.

Three of these unplanned releases were known of at the time the 200-BP-1 Phase I RI work plan (DOE-RL 1990a) was prepared. The fourth unplanned release (UN-200-E-89) was designated by DOE as such in 1991. Interim stabilization measures were undertaken at the unplanned release and completed in 1991 to address the surface radiological contamination until a final remediation strategy is implemented (Hayward 1992).

Available information (including the date of the release and actions taken) regarding the details of these UNs is provided in Appendix H.

UN-200-E-63 was an unplanned mixed waste release, and is described as tumbleweeds becoming contaminated by uptake of radionuclides from the BC crib and trench. This vegetation was then uprooted and blown around contaminating the surrounding ground surface. The contaminated vegetation was removed and a weed control program was initiated to control future growth of tumbleweeds.

Waste unit UN-200-E-9 involved approximately 41,635 l (11,000 gl) of TBP supernatant waste which leaked onto the ground from the 216-B crib flush tank (Figure 3-1). The spill occurred in an area directly north of the flush tank. Most of the wastes were removed to a site south of the 216-B-43 crib and were covered with 0.6 m (2 ft) of clean soil. The contamination left near the flush tank was covered with 3 m (10 ft) of clean soil (DOE-RL 1992a).

Surface radiological conditions at the UN-200-E-89 area were documented in a series of surveillance reports identified in Hayward (1992). The reports identified soil surface

radiation readings that exceeded allowable levels, and the lack of an adequate surface barrier to prevent migration. This unplanned release was originally named UN-216-E-17 which has been changed to UN-200-E-89. The interim stabilization action was undertaken to correct the deficiencies identified in these reports and bring the site into compliance with WHC-CM-7-5, Part L, "Inactive Radioactive Waste Sites" (WHC 1988a). The source of the contamination was suspected to be the manhole and risers near the 216-B cribs, and the BX/BY Tank farms (Hayward 1992).

Waste unit UN-200-E-89 was bounded on the east by Baltimore Avenue, on the north by 12th Street, and on the south by the BY Tank Farm (Figure 3-1 and 3-3). The majority of the unplanned release (approximately 4.3 ha [10.6 ac]) was within the 200-BP-1 operable unit. The remainder (approximately 0.85 ha [2.1 ac]) was within the 200-BP-7 operable unit to the south (Hayward 1992).

Interim stabilization actions consisted of a combination of scraping and re-placement of surface contaminated soils followed by covering (stabilization) with clean soil and rock, and covering (stabilization) of soils in-place. The UN-200-E-89 area was divided into 7 zones based on the treatment they would receive for the interim stabilization. Scraping occurred to depths of up to 15 cm (6 in.) over much of the UN-200-E-89 area. Scraped soils were placed either in the low area in the center of 216-B-43 through -50 or over the surface of 216-B-57. Once the consolidation was complete, the crib areas were stabilized with approximately 46 to 61 cm (18 to 24 in.) of clean soil. Figure 2-3 depicts the zones within the UN-200-E-89 area and treatment received. Additional detailed information of the UN-200-E-89 interim stabilization activities are included in Hayward (1992).

Waste unit UN-200-E-110 involved first-cycle waste from the 112-BY tank in the 241-BY Tank Farm and impacted an area of approximately 2,320 m^2 (25,000 ft^2) around the 112-BY pit. It is possible, although currently unknown, that the release flowed into the 200-BP-1 operable unit. No information was obtained on the quantity of the release or whether remedial action was taken.

3.1.3 Interactions with Other Operable Units

As illustrated in Figure 3-2, the 200-BP-1 operable unit is bordered by operable units 200-BP-4 to the east, 200-BP-7 and 200-BP-3 to the south, and the 200-BP-10 operable unit to the west. The 600 Area, which includes all areas at the Hanford Site not located in the other designated areas (i.e., 100, 200, 300, 400, and 1100), borders the operable unit to the north.

Operable units adjacent to 200-BP-1 contain waste storage, waste burial, or waste infiltration systems. The 200-BP-4 and 200-BP-3 operable units contain cribs. The 200-BP-7 contains the 241-BY, 241-BX, and 241-B Tank Farms. The 200-BP-10 contains burial grounds for solid waste disposal. The 600 Area immediately to the north of 200-BP-1 does not contain process facilities or waste disposal systems.

The waste disposal cribs and waste storage tanks in adjacent operable units received or contain many of the same constituents as the cribs in 200-BP-1. Cribs in adjacent operable units were designed for underground infiltration of waste water effluents. Leaks have occurred from single-shell storage tanks in 200-BP-7, resulting in release of wastes

containing higher concentrations of radionuclides than were normally disposed of in the cribs (WHC 1988).

Most of these operable units are hydraulically upgradient of 200-BP-1 and may be contributing to groundwater contamination in the area (Section 3.6.2.2). If the groundwater beneath or near the 200-BP-1 operable unit is found to have contributing contamination from other operable units, any remedial action taken for the contaminated groundwater must include the contamination emanating from the other operable units as well as the contribution from the sources within 200-BP-1.

3.2 METEOROLOGICAL CHARACTERISTICS

This section presents an interpretation of meteorological data for the Hanford Site and the 200-BP-1 operable unit. The data has been collected primarily at the Hanford Meteorologic Station (HMS), which is located between the 200 East and 200 West Areas of the Hanford Site, approximately 8 km (5 mi) to the west of the 200-BP-1 operable unit. Data from the HMS are assumed to be representative of the general climatic conditions of the region, and of the 200-BP-1 operable unit.

A large compilation and summary of HMS data for the period from 1946 to 1980, and of additional data from other nearby stations for the period 1912 to 1943, was conducted by Stone et al. (1983). The discussion which follows, unless otherwise noted, is based on this compilation. The location of the HMS, as well as of the other meteorologic monitoring locations on the Hanford Site, are shown in Figure 3-3.

3.2.1 Precipitation

The Cascade Range is located approximately 130 km (80 mi) west of the Hanford Site and has an average crest elevation of about 1,800 m (6,000 ft) AMSL. This mountain range creates a rain shadow that limits the average total annual precipitation at the HMS to about 16 cm (6.3 in.). The total annual precipitation (98 percentile) ranges from 8 to 27.9 cm (3.15 to 11 in.). The three months November through January generally contribute approximately 42% of this total, while the three months July through September contribute only 12%. January is the wettest month with an average of 2.3 cm (0.92 in.) while July is the driest month with an average of only 0.38 cm (0.15 in.). Monthly average precipitation amounts at HMS are shown in Figure 3-4. Precipitation intensity is greatest in the summer months. This seasonal intensity peak coincides with the thunderstorm season.

Data on the expected frequency of precipitation intensity and short-period duration (24 h or less) are presented in Figure 3-5 and Table 3-2. The information, which is for the HMS, is based on data collected for the period 1947 through 1969.

Rain is the usual form of precipitation at the HMS, but snowfall regularly occurs during winter, and hail storms, though infrequent, may occur during the summer thunderstorm season. Approximately 38% of all precipitation during the months of December through February is in the form of snow. However, in only one winter in four does an accumulation in excess of 15.2 cm (6 in.) occur. The average annual snowfall is 33 cm (13.2 in.). Complete snowmelt generally occurs within a month of a snowstorm.

3.2.2 Temperature and Humidity

The summer months at the Hanford Site are typically hot and dry, and winters are moderately cold. July is the warmest month of the year with an average temperature of 24.7 °C (76.4 °F), and January is the coolest month with an average temperature of -1.5 °C (29.3 °F). Average high temperatures in the summer are 37 °C (100 °F) and average lows in winter are -5 °C (23 °F). Annual high temperatures are normally recorded in July and the annual lows normally occur in January. Historical extreme temperature readings of 46 °C (115 °F) and -29 °C (-20 °F) have been recorded. Monthly mean temperatures for the period 1912 through 1980, as collected at the Hanford Townsite and the HMS, are depicted in Figure 3-6.

The diurnal temperature range is substantial, due to low humidity. During summer months, when the average relative humidity is 30 to 40%, the diurnal temperature range is greatest, on the order of 15 °C (27 °F). In winter, with relative humidity ranging from 60 to 80%, the diurnal temperature range is reduced to about 8 °C (14 °F) (DOE-RL 1990b). Figure 3-7 depicts the monthly average high and low temperatures for the period 1951 to 1980.

The annual average relative humidity at the HMS is 54%, with maxima during the winter months (averaging about 75%) and minimum average relative humidity during the summer (about 35%) (Stone et al. 1983). Monthly mean relative humidity for the period 1950 through 1980 is depicted in Figure 3-6.

3.2.3 Wind

Wind directions at the HMS varies over 360 degrees, with a prevailing wind direction from west-northwest to northwest for every month of the year (average of 31.6% of the time). Secondary maxima occur for southwesterly winds. The months of June and July have the highest percentage of winds from the WNW and NW (38 and 37%, respectively). October has the lowest percentage (25%) from those directions. Monthly wind roses for the HMS are shown in Figure 3-8.

Elsewhere on the Hanford Site, the predominant wind direction may differ from those at HMS, as shown in Figure 3-9. Mountain ridges and river valleys locally influence wind direction, particularly along the Columbia River where predominant wind directions parallel the river. There is also a strong diurnal effect observed from March through August, when wind speeds tend to increase 7 to 10 km/h (4 to 6 mph) during the afternoon and evening hours (DOE-RL 1990b).

Monthly average wind speeds are generally lowest during the winter months, averaging 10 to 11.7 km/h (6 to 7 mph), and highest during the summer, averaging 15 to 16.7 km/h (9 to 10 mph). The highest monthly average wind speeds occur in June (15.3 km/h [9.2 mph]) and the lowest monthly average wind speeds occur in November and December (10.2 km/h [6.1 mph]).

At the HMS, the strongest winds observed, with speeds up to 130 km/h (80 mph), generally are southwesterly. Most hourly wind speeds greater than 52 km/h (31 mph) are from the south-southwest to west-southwest and occur at the highest frequency from March through May (Hulstrom 1992).

The annual frequency distribution of the near-surface wind direction and wind speed in the 200 East Area is presented numerically in Table 3-3. In the table, the frequency of occurrence of each combination of wind direction sector and wind speed is presented. Data used to create the frequency distribution are from observations made at the 200 East Area meteorological station for the period 1979 to 1988 and represent hourly averages. Median hourly average wind speed at the 200 East Area is about 7 km/h (4 mph).

Wind-blown dust accompanies strong winds on the Hanford Site. Blowing dust originating from the site itself has been observed at wind speeds greater than 32 km/h (19 mph). Dust entrained elsewhere and transported to the Hanford Site has been observed for lower wind speeds of 7 km/h (4 mph) (DOE-RL 1990b). Observations of blowing dust may occur with any wind direction, however, the strongest winds at the HMS are from the southwest and therefore there are more cases of blowing dust from that direction. Dust transported to the Hanford Site from elsewhere is most often associated with winds from the north and northeast.

3.2.4 Evapotranspiration

Pan evaporation data was obtained from the Washington State University Cooperative Extension for Prosser, WA located approximately 42 km (25 mi) southwest of the 200 East Area (Appendix G). Monthly rates of pan evaporation at the Washington State University Irrigated Agriculture Research and Extension Center (IAREC) average from about 8.1 to 25.4 cm (3.2 to 10 in.). These averages are based upon data collected over the period 1924 to 1988 for the months April through October. Total pan evaporation over the April through October period averaged about 126.6 cm (49.9 in.). This seasonal component represents approximately 80% of the total annual pan evaporation. Average monthly pan evaporation at Prosser is depicted in Figure 3-10.

Free surface evaporation (or potential evaporation) is expected to equal approximately 70% of the pan evaporation for the Hanford Site vicinity, or about 111 cm (43 in.) (Weather Bureau 1966). Free water surface evaporation is of interest because it closely represents the potential evaporation from adequately watered surfaces, such as vegetation and soil, and the evaporation from a surface body of water.

For the Hanford Site a monitoring program was conducted, beginning in the late 1970s, to study groundwater recharge and measure parameters that affect recharge rates. Rockhold et al. (1990) reported on water balance data which was collected as part of this program from three sites in 1988 and 1989. The sites included the 300 Area buried waste test facility and grass site, and the 200 East Area closed-bottom lysimeter. While evapotranspiration was not specifically reported for the 200 East Area site, it was reported that measured water contents in the soil implied that significant recharge had not occurred within the lysimeter.

For the 300 Area buried waste test facility, evaporation and transpiration were determined to be about 14.3 cm (5.6 in.) for a bare surface and 19.9 cm (7.9 in.) for a vegetated surface, using measurements of changes in water storage, drainage, and precipitation. Precipitation during this period was approximately 18 cm (7.1 in.). Drainage was about 4 cm (1.6 in.) from the bare surface and 1 cm (0.4 in.) from the vegetated surface. The excess of evapotranspiration and drainage over precipitation was compensated for by a reduction in soil moisture.

Figure 3-11 presents a plot of monthly evapotranspiration totals for the north (bare) and south (vegetated) weighing lysimeters at the buried waste test facility during the period December 1987 to August 1990. This figure illustrates the large seasonal and annual variations in evapotranspiration and the large differences that can occur as a result of vegetation.

3.3 SURFACE HYDROLOGICAL CHARACTERISTICS

This section provides a characterization of surface water hydrology, regionally within the Pasco Basin and locally in the vicinity of the 200-BP-1 operable unit. The regional information is presented with attention focused on those aspects which are felt to relate directly to the 200-BP-1 operable unit Phase I RI. Additional information on the regional hydrology may be found in DOE (1988), ERDA (1975) and Skaggs and Walters (1981).

3.3.1 Regional Surface Hydrology

Primary surface water features associated with the Hanford Site are the Columbia River and its major tributaries, the Yakima, Snake and Walla Walla rivers, as well as ephemeral streams, a natural pond, and man-made ditches and ponds associated with nuclear fuel processing and waste disposal activities. No perennial streams originate within the Pasco Basin. The major water bodies are shown in Figure 3-12. The locations of major drainage divides are depicted in Figure 3-13. Physical aspects of these surface water bodies are discussed below.

3.3.1.1 Major Rivers. The major surface water body in the Pasco Basin is the Columbia River, which flows from the Canadian Rocky Mountains through Washington State, and along the Oregon border, to the Pacific Ocean. Enroute to the Pacific, the Columbia River crosses the northern portion of the Hanford Site (approximately 20 km [12 mi] to the northwest of the 200-BP-1 operable unit), then turns southward to form the Hanford Site's eastern boundary. About two-thirds of the Hanford Site drains into the Columbia River; the remaining one-third (in the western and southern portions of the Hanford Site) drains into the Yakima River (Figure 3-13). Both the Yakima and the Columbia rivers are important sources of water for domestic, agricultural, industrial, and recreational users in the Pasco Basin (DOE 1987; Jaquish and Bryce 1990).

Flow of the Columbia River is regulated by 11 dams within the United States, 7 upstream and 4 downstream of the Hanford Site (PNL 1988a). Priest Rapids Dam is the nearest impoundment upstream of the site, and McNary Dam is the nearest dam downstream.

The Hanford Reach of the Columbia River extends from Priest Rapids Dam, approximately 8.5 km (5.3 mi) above the Hanford Site boundary, to the head of Lake Wallula approximately at the southeastern Hanford Site boundary. Lake Wallula is created by McNary Dam. The Hanford Reach, which is approximately 100 km (60 mi) in length, is the last unimpounded segment of the Columbia River in Washington State and its shoreline remains largely undeveloped (Jaquish and Bryce 1990). Several drains and intakes are present along this reach, including irrigation outfalls from the Columbia Basin Irrigation Project, the Washington Public Power Supply System Nuclear Project 2, and the Hanford Site intakes for onsite water use.

Volumetric flow rates in the Columbia River along the Hanford Reach vary widely and erratically due to operations of the Priest Rapids Dam, operated by Public Utility District No. 2 of Grant County, and the operational practices of the nearby upstream dams. A minimum flow rate of 1,000 m^3/s (36,000 ft^3/s) has been established at Priest Rapids (PNL 1988a).

The average daily flow varies from a high of approximately 8,000 m^3/s (283,000 ft^3/s) in June to a low of about 2,000 m^3/s (70,000 ft^3/s) in October and November. The average daily flow over the entire period of record is approximately 3,400 m^3/s (119,000 ft^3/s). Monthly average flows have ranged as high as 16,700 m^3/s (590,000 ft^3/s) which occurred in the month of June to about 600 m^3/s (21,000 ft^3/s) for January and February.

The flow duration curve for the Columbia River below Priest Rapids Dam is depicted in Figure 3-14. This curve depicts the probability of exceeding a given daily mean discharge based upon data collected over the period 1939 to 1979.

River stage measurements taken in the 100-H Area are presented in Figure 3-15. The figure, which is for the period September 1991 to July 1992, depicts the typical seasonal variation in river stage exhibited over an approximately one year period. As seen in the figure, the surface elevation of the river varies from a maximum of about 116 m (381 ft) AMSL to a minimum of about 112 m (369 ft) AMSL. Typical daily fluctuations are on the order of 1.2 to 1.5 m (4 to 5 ft), reflecting the discharge control related to power generation needs at upstream dams. These river fluctuations cause fluctuations in groundwater levels, with the magnitude of this effect being most pronounced within about 300 m (1,000 ft) of the river bank. In the vicinity of the river, the water table fluctuations result in a highly variable gradient and flow pattern (WHC 1991).

3.3.1.2 Other Naturally-Occurring Surface Waters. No perennial streams occur within the central portion of the Hanford Site. Cold Creek and its tributary, Dry Creek, are part of the Yakima River watershed and originate in the synclinal valleys west of the Hanford Site (Figures 3-15 and 3-16). Both streams receive some base flow from springs along portions of their reaches. Other reaches are ephemeral, responding to seasonal runoff from precipitation and snowmelt.

The Cold Creek drainage ultimately connects to the Yakima River about 2 km (1 mi) upstream from Horn Rapids Dam (Figure 3-13). Actual flow in Cold Creek and Dry Creek, which results from precipitation onto Rattlesnake Hills, Umtanum Ridge, and Yakima Ridge, is not well documented; however, flood magnitudes in Cold Creek, having recurrence intervals of 5 and 10 years, were estimated to be 60 and 120 m^3/s (2,100 and 4,400 ft^3/s), respectively, in the creek's lower reaches (Skaggs and Walters 1981).

West Lake, located about 2.8 km (1.7 mi) north of the 200 East Area, is a shallow pond, with an average depth of about 1 m (3 ft) and a surface area of approximately 4 ha (10 ac) (Fuchs et al. 1985). The pond has previously been described as the "only naturally occurring pond on the Hanford Site" (DOE 1988; DOE-RL 1990c; DOE-RL 1990a). This statement is valid in the sense that the pond does not consist of a disposal pond built and constructed specifically as part of the Hanford Site operations. However, the source of recharge to the lake is groundwater which is locally mounded due to infiltration resulting from the 200 Areas operations (Graham 1983). The pond represents a topographic depression which intersects the elevated water table surface. The rise in the water table which has occurred as a result of the 200 Areas operations may have directly resulted in

the occurrence of the lake. The pond therefore represents a natural or artificially derived surface water body or wetland.

3.3.1.3 Man-Made Ditches and Ponds. On the Hanford Site, wastewater discharge into ponds and ditches occurs in the 200 and 300 Areas (Figure 3-12). At these locations, several ponds and ditches exist to hold waste waters, which eventually evaporate or infiltrate.

3.3.2 Local Surface Water Hydrology

There are no perennial or ephemeral streams in the 200 East Area or adjacent land. Ephemeral surface runoff will follow general drainage patterns. Surface drainage from the 200-BP-1 operable unit is primarily to the north (Figure 2-4) and will be directed toward a topographic depression located approximately 0.7 km (0.4 mi) north of the operable unit boundary.

Surface drainage onto the 200-BP-1 operable unit is from the BY Tank Farm immediately to the south and from areas to the southwest. The BY Tank Farm is situated on a flat bench constructed higher in elevation than, and sloping toward, the 200-BP-1 operable unit. Surface topography of the 200-BP-1 operable unit is depicted in Figure 2-4.

3.4 GEOLOGICAL CHARACTERISTICS

This section provides a description of the regional and local geologic characteristics of the 200-BP-1 operable unit. The regional information has been largely summarized from a number of pre-existing technical documents which address the geologic conditions of the Hanford Site and the 200 East Area. These include DOE (1988), Delaney et al. (1991), and Lindsey et al. (1992) among others. The description of geologic conditions local to the 200-BP-1 operable unit is based upon these previous sources, as well as on work which was undertaken in the 200-BP-1 operable unit Phase I RI, specifically under Tasks 2, 4, and 6 of the 200-BP-1 operable unit work plan (DOE-RL 1990a). The results of the geologic investigations performed under these tasks are presented in Hoffman (1992), Hoffman et al. (1992), Swanson (1992), and Connelly et al. (1992).

3.4.1 Topography and Physiography

The Hanford Site is situated within the Pasco Basin, one of a number of topographic and structural depressions located within the Columbia Plateau physiographic province, a broad basin located between the Cascade Range and Rocky Mountains (Delaney et al. 1991). The Pasco Basin is bounded on the north by the Saddle Mountains; on the west by Umtanum Ridge, Yakima Ridge, and the Rattlesnake Hills; and on the east by the Palouse slope. Topography of the Hanford Site is depicted in Figure 3-16.

The Hanford Site includes about 900 km^2 (350 mi^2) of terrace lands located south and west of the Columbia River within the semiarid Pasco Basin of south-central Washington. The terrace plains rise gradually north and west from an altitude of about 104 m (340 ft) at Richland to 213 to 244 m (700 to 800 ft) in the northwestern part of the site. From these high terraces the surface descends to the 137-m (450-ft) terraces along the river. Toward the west the terrace lands terminate against the slopes and interridge valleys of low linear

mountains known collectively as the Yakima Ridges. Rattlesnake Mountain, at the southwest edge of the site, rises to an elevation of 1,085 m (3,560 ft). A few bedrock outliers, such as Gable Mountain, outcrop above the terraces of the Hanford Site (Newcomb et al. 1972).

Surface topography at the site is the result of (1) uplift of anticlinal ridges, (2) Pleistocene cataclysmic flooding, (3) Holocene eolian activity, and (4) landsliding (DOE 1988). Uplift of the ridges began in the Miocene epoch (<17 million years) and continues to the present. Cataclysmic flooding occurred when ice dams in western Montana and northern Idaho were breached, allowing large volumes of water to spill out across eastern and central Washington (Baker et al. 1991). Flooding began approximately 750 thousand years and ended approximately 13 thousand years, during the Late Pleistocene Epoch. These floods left behind an array of unique landforms around the Hanford Site including anastomosing flood channels, giant current ripples, bergmounds, and giant flood bars. Since the end of the Pleistocene Epoch, winds have locally reworked the fine-grained flood sediments, depositing sand dunes in the lower elevations and loess (windblown silt) around the margins of the Pasco Basin. Generally, sand dunes have been stabilized by vegetation except where they have been reactivated where vegetation is disturbed.

The 200 Areas are situated on a broad flat terrace called the 200 Areas (includes the 200 East and 200 West Areas) plateau located near the center of the Hanford Site at an elevation of approximately 198 to 229 m (650 to 750 ft) AMSL. The plateau decreases in elevation to the north and east toward the Columbia River. Decreases in elevation to the north and northeast of the 200-BP-1 operable unit occur primarily along the edges of terraces. The terrace escarpments are steep, with elevation changes between 15 and 30 m (50 and 100 ft).

3.4.2 Regional Geology

A summary of the regional geologic characteristics of the Pasco Basin and the Hanford Site is presented below in terms of stratigraphy, and structure. Regional conditions of the area have been described in detail in a number of earlier studies and technical papers, including DOE (1988), Delaney et al. (1991) and Lindsey et al. (1992).

3.4.2.1 Regional Stratigraphy. The Hanford Site lies within the Pasco Basin, a regional structural and topographic, sediment-filled depression. The sediments of the Pasco Basin are underlain by Miocene-age basalt of the Columbia River Basalt Group, a thick sequence of flood basalts that covers a large area in eastern Washington, western Idaho and northeastern Oregon. The sediments overlying the basalts, from oldest to youngest, include: the Miocene-Pliocene Ringold Formation, local alluvial deposits of possible Late Pliocene or probable Early Pleistocene age, local "Palouse" soil of mostly eolian origin, glaciofluvial deposits of the Pleistocene Hanford formation, and surficial Holocene eolian and fluvial sediments. The generalized stratigraphy of the Hanford Site is depicted in Figure 3-17 and described below from oldest to youngest, in the order of their deposition.

3.4.2.1.1 Columbia River Basalt Group and the Ellensburg Formation. The Columbia River Basalt Group consists of an assemblage of tholeiitic, continental flood basalts of Miocene Age with accumulated thickness in excess of 3,000 m (10,000 ft) within the Pasco Basin. These flows cover an area of more than 163,700 km^2 (63,000 mi^2) in

Washington, Oregon, and Idaho and have an estimated volume of about 174,356 km^3 (40,800 mi^3). The majority of the flows were erupted 14.5 to 17 Ma (DOE 1988).

The Columbia River Basalt Group is formally divided into five formations (from oldest to youngest): Imnaha Basalt, Picture Gorge Basalt, Grande Ronde Basalt, Wanapum Basalt, and Saddle Mountains Basalt. Of these, all are present within the Pasco Basin except for the Picture Gorge Basalt. The Saddle Mountains Basalt, divided into the Ice Harbor, Elephant Mountain, Pomona, Esquatzel, Asotin, Wilbur Creek, and Umatilla Members (Figure 3-17), forms the uppermost basalt unit throughout most of the Pasco Basin. The Elephant Mountain Member is the uppermost unit beneath most of the Hanford Site except near the 300 Area where the Ice Harbor Member is found and north of the 200 Areas where the Saddle Mountains Basalt has been eroded down to the Umatilla Member in the Gable Gap area (Delaney et al. 1991). The Elephant Mountain Member has also been locally eroded down to the Rattlesnake Ridge interbed, approximately one mile north of the 200-BP-1 operable units and in the vicinity of the northeast corner of the 200 East Area. On anticlinal ridges bounding the Pasco Basin, erosion has removed the Saddle Mountains Basalt, exposing the Wanapum and Grande Ronde basalts.

The Ellensburg Formation consists of all sedimentary units that occur between the basalt flows of the Columbia River Basalt Group in the central Columbia Basin (Reidel and Fecht 1981) (Figure 3-17). The Ellensburg Formation generally consists of two main lithologies: volcaniclastics and siliciclastics. The volcaniclastics consist mainly of primary pyroclastic air fall deposits and reworked epiclastics derived from volcanic terrains west of the Columbia Plateau. Siliciclastic strata consist of clastic, plutonic, and metamorphic detritus derived from the Rocky Mountain terrain. The stratigraphic names for the individual units of the Ellensburg Formation applicable to the Hanford Site are shown in Figure 3-17.

At the Hanford Site the three uppermost units of the Ellensburg Formation are the Levey interbed, the Rattlesnake Ridge interbed, and the Selah interbed. The Levey interbed is confined to the vicinity of the 300 Area. The Rattlesnake Ridge and Selah interbeds are found beneath most of the Hanford Site (Lindsey et al. 1992).

3.4.2.1.2 Suprabasalt Sediments. The suprabasalt sedimentary sequence at the Hanford Site is up to approximately 230 m (750 ft) thick in the west-central Cold Creek syncline, while it pinches out against the anticlinal ridges that bound or are present within the Pasco Basin. The suprabasalt sediments are dominated by laterally extensive deposits of the late Miocene to Pliocene-age Ringold Formation and the Pleistocene-age Hanford formation. Locally occurring strata separating the Ringold and Hanford formations are assigned to the informally defined Plio-Pleistocene unit, early "Palouse" soil, and pre-Missoula gravels comprise the remainder of the sequence.

Ringold Formation. Overlying the Columbia River Basalt Group is the late Miocene to Pliocene-age Ringold Formation (Fecht et al. 1987, DOE 1988). The Ringold Formation accumulated to thicknesses of up to 365 m (1,200 ft) in the Pasco Basin. On the Hanford Site, the Ringold Formation is up to 185 m (600 ft) thick in the deepest part of the Cold Creek syncline south of the 200 West Area and 170 m (560 ft) thick in the western Wahluke syncline near the 100-B Area. The Ringold Formation pinches out against the anticlinal flanks that bound or are present within the Pasco Basin, and is largely absent in the northern and northeastern parts of the 200 East Area and adjacent areas to the north (Delaney et al. 1991, Lindsey et al. 1992).

Post-Ringold Pre-Hanford Sediments. Thin alluvial deposits situated stratigraphically between the Ringold Formation and Hanford formation are found within the Pasco Basin. The three informally defined units include: (1) the Plio-Pleistocene unit; (2) the early "Palouse" soil; and (3) the Pre-Missoula gravels. The Plio-Pleistocene unit and early "Palouse" soil are not found in or near the 200 East Area. They are found to the west of the study area near the eastern boundary of the 200 West Area. The pre-Missoula gravels are not found in the study area. Because of the general absence of these units from the study area, they will not be discussed further. The Plio-Pleistocene unit and early "Palouse" soil are described in detail in Last et al. (1989) and Lindsey et al. (1991). The pre-Missoula gravels are discussed in PSPL (1982) and Fecht et al. (1987).

Hanford formation. The informally designated Hanford formation consists of unconsolidated, glaciofluvial sediments that were deposited during several episodes of cataclysmic flooding during the Pleistocene Epoch. The sediments are composed of pebble to boulder gravel, fine- to coarse-grained sand, and silt. These sediments are divided into three facies: (1) gravel dominated, (2) sand-dominated, and (3) silt-dominated (Lindsey et al. 1992). These facies are referred to as coarse-grained deposits, plane-laminated sand facies, and rhythmite facies, respectively (Baker et al. 1991). The silt-dominated deposits are also referred to as "Touchet" beds, and the gravel-dominated facies generally correspond to the Pasco gravels.

The Hanford formation is thickest in the vicinity of the 200 Areas where it is up to 107 m (350 ft) thick (Lindsey et al. 1992). The formation was deposited by cataclysmic flood waters that flowed out of glacial lake Missoula (Fecht et al. 1987; DOE 1988; and Baker et al. 1991). The deposits are absent from ridges above approximately 360 m (1,180 ft) AMSL, the highest level of cataclysmic flooding in the Pasco Basin (Delaney et al. 1991).

Holocene Surficial Deposits. Holocene surficial deposits consist of silt, sand, and gravel that form a (<4.9 m [<16 ft]) veneer across much of the Hanford Site. These sediments were deposited by a mix of eolian and alluvial processes.

3.4.2.2 Regional Geologic Structure. The Hanford Site is located within the Pasco Basin near the eastern edge of the Yakima Fold Belt. The Yakima Fold Belt consists of a series of segmented, narrow, asymmetric, east-west trending anticlines separated by broad synclines or basins that, in many cases, contain thick accumulations of Neogene- to Quaternary-aged sediments (DOE 1988; Smith et al. 1989). The Pasco Basin is one of the larger structural basins of the fold belt.

The northern limbs of the anticlines of the Yakima Fold Belt generally dip steeply to the north, or are vertical. The southern limbs generally dip at relatively shallow angles to the south. Thrust or high-angle reverse faults with fault planes that strike parallel or subparallel to the axial trends are principally found on the north sides of the anticlines. The amount of vertical stratigraphic offset associated with these faults varies.

Deformation of the Yakima Folds occurred under north-south compression and was contemporaneous with the eruption of the basalt flows. The fold belt was enlarging during the eruption of the Columbia River Basalt Group and continued to enlarge through the Pliocene, into the Pleistocene, and perhaps to the present.

The Pasco Basin is a structural depression bounded on the north by the Saddle Mountain anticline, on the west by the Umtanum Ridge, Yakima Ridge, and Rattlesnake

Hills anticlines, and on the south by the Rattlesnake Mountain anticline (Figure 3-18). The Palouse slope, a west-dipping monocline, bounds the Pasco Basin on the east. The Pasco Basin is divided into the Wahluke and Cold Creek synclines by the Gable Mountain anticline, the eastern extension of the Umtanum Ridge anticline.

The Cold Creek syncline (Figure 3-18) lies between the Umtanum Ridge-Gable Mountain uplift and the Yakima Ridge uplift, and is an asymmetric and relatively flat-bottomed structure. The bedrock of the northern limb dips gently to the south, and the southern limb dips steeply to the north. The deepest parts of the Cold Creek syncline, the Wye Barricade depression and the Cold Creek depression, are located approximately 12.5 km (7.5 mi) southeast of the 200 Areas and just to the west-southwest of the 200 West Area, respectively.

3.4.3 Local Geology

This section focuses on the geologic characteristics of the 200-BP-1 operable unit and vicinity. Information presented has been compiled from a variety of sources, including pre-existing technical reports and documents of 200 East Area, as well as the results of the recent field investigative work undertaken for the 200-BP-1 operable unit Phase I RI, specifically under Tasks 2, 4, and 6 (DOE-RL 1990a). The results of the geologic investigations performed under these tasks are presented in Hoffman (1992), Hoffman et al. (1992), Swanson (1992), and Connelly et al. (1992).

The discussion will focus on the characterization of geologic conditions within the context of a defined study area. The study area discussed in this report includes the 200 East Area, and extends northward to the Gable Gap area. The outline of the study area is defined to include the anticipated areal extent of groundwater contamination, based upon the existing information, which has resulted from 200-BP-1 operable unit and nearby related activities.

3.4.3.1 Topography and Geomorphic Setting. The topography in the vicinity of the 200-BP-1 operable unit was formed primarily by Pleistocene cataclysmic floods beginning at least 750 Ka and ending approximately 13 Ka (Baker et al. 1991). These floods left behind an array of unique landforms including anastomosing flood channels, giant current ripples, bergmounds, and giant flood bars. The 200-BP-1 operable unit is situated at an elevation of approximately 192 m (630 ft) AMSL on one of these landforms, the Cold Creek Bar (Bretz et al. 1956). This flood bar is a compound bar built by multiple floods (DOE 1988). During flooding it prograded southward to its present position. The northern part of the bar has undergone erosion by flood waters receding from the basin, resulting in the creation of at least four major channels, as well as additional minor channels, that have been recognized near the Gable Mountain, Gable Butte area (Fecht 1978).

Eolian activity in the Columbia Basin is a major geomorphic agent. However, in the vicinity of the 200-BP-1 operable unit, wind has done little but to locally rework and redistribute flood deposits. This has produced sheet sands which blanket the surface topography (Hoffman et al. 1992). The surface topography of the 200-BP-1 operable unit is depicted in Figure 2-4.

3.4.3.2 Local Stratigraphy. The geologic units of interest found in the vicinity of the 200-BP-1 operable unit include from oldest to youngest: (1) the Pomona Member of the Saddle Mountains Basalt, (2) the Rattlesnake Ridge interbed of the Ellensburg Formation, (3) the Elephant Mountain Member of the Saddle Mountains basalt, (4) the Ringold Formation, (5) the Hanford formation, and (6) the Holocene surficial deposits. The Ringold Formation, while not present immediately beneath the 200-BP-1 operable unit or to the north, is present in the operable unit vicinity, and as such is included in the local geology discussion. The stratigraphic relationships of these units below the 200-BP-1 operable unit are shown in a generalized stratigraphic column (Figure 3-19), a series of cross-sections (Figures 3-20 to 3-24), a fence diagram (Figure 3-25), and isopach and structure contour maps (Figures 3-26 to 3-31).

3.4.3.2.1 Pomona Member. The older of the two basalt units discussed here, the Pomona Member, consists of a single flow that was erupted approximately 12 Ma (McKee et al. 1977). It is one of the most extensive and voluminous Saddle Mountains basalt flows. The presence of the unit and thickness over the study area is depicted in Figure 3-26. As seen in the figure, the thickness of the Pomona Member in the study area varies over a relatively narrow range from about 50 to 60 m (164 to 197 ft). The unit thickens slightly to the south (Graham et al. 1984). The Pomona Member is overlain by the Rattlesnake Ridge interbed of the Ellensburg Formation in the study area. Along the axis of the Umtanum Ridge-Gable Mountain anticline, northwest of West Lake (the Gable Gap area), the unit has been completely removed by floodwater erosion (Figure 3-26) (Graham et al. 1984).

3.4.3.2.2 Rattlesnake Ridge Interbed. The Rattlesnake Ridge interbed of the Ellensburg Formation is bounded on top by the Elephant Mountain Member and on the bottom by the Pomona Member, both of the Saddle Mountains Basalt (Reidel and Fecht 1981). The interbed was deposited by the ancestral Columbia River subsequent to Pomona volcanism and prior to the extrusion of the Elephant Mountain Member (Hoffman et al. 1992).

The interbed consists primarily of air fall and fluvially reworked fine-grained siliciclastic material, and micaceous-arkosic sands derived from the Rocky Mountain terrain (Fecht 1978). These sands typically display an orange to tan color, medium-grained texture, and are locally cross-bedded in outcrop. Light grey, silty, vitric tuff typically overlies the sands (Fecht 1978).

As depicted in Figure 3-27, the interbed in the study area generally thickens to the south from a maximum thickness of over 25 m (82 ft) in the southwest portion of the 200 East Area to 0 m (0 ft) in the Gable Gap Area where the interbed has been completely removed by floodwater erosion (Graham et al. 1984; Fecht 1978). In the immediately vicinity of the 200-BP-1 operable unit (borehole 299-E33-12), the unit is approximately 20 m (65 ft) thick. From the 200-BP-1 operable unit to the north, the thickness of the unit is generally from 10 to 20 m (33 to 66 ft).

North of the 200 East Area, in the vicinity of boreholes 699-53-55 and 699-55-55, the Rattlesnake Ridge interbed is in direct contact with the overlying Hanford formation due to complete erosion of the overlaying Elephant Mountain basalt (Figures 3-21 and 3-22). In this "erosional window" feature shown in Figure 3-21, the Rattlesnake Ridge has been partially eroded, as evidenced by the relative thinness of the interbed at borehole 699-53-55 (11 m [36 ft]). As stated above, the unit is absent in the Gap Area where erosion has

occurred along the crest of the Umtanum Ridge-Gable Mountain anticline (Fecht 1978; Graham et al. 1984).

3.4.3.2.3 Elephant Mountain Member. The uppermost basalt in the vicinity of the 200-BP-1 operable unit is the Elephant Mountain Basalt Member of the Saddle Mountains Basalt (Reidel and Fecht 1981). The Elephant Mountain Member consists of two tholeiitic, medium-to-fine grained basalt flows: the upper Ward Gap flow and the lower Elephant Mountain flow. The flows were erupted approximately 10.5 Ma (McKee et al. 1977). Only the Elephant Mountain flow is present in the study area (Hoffman et al. 1992). The Elephant Mountain Member thickens from approximately 21 m (69 ft) in the northwestern portion of the study area to over 30 m (100 ft) in the south (Lindsey et al. 1992). Figure 3-28 depicts the thickness of the unit within the study area.

The elevation of the top of basalt across the study area is depicted in Figure 3-29. As shown in the figure, the study area is located on the south flank of a major fold - the Umtanum Ridge-Gable Mountain anticline. Beneath the 200-BP-1 operable unit, the basalt surface dips generally to the south-southwest from the axis of the anticline towards the axis of the Cold Creek syncline.

The buried surface of the Elephant Mountain basalt has been eroded to various degrees in the study area (Fecht 1978; DOE 1988; Connelly et al. 1992; Delaney et al. 1991; Lindsey et al. 1992; and Hoffman et al. 1992). North of the 200-BP-1 operable unit in the Gable Gap area, the Elephant Mountain basalt, as well as the Rattlesnake Ridge interbed and Pomona basalt, has been totally eroded, exposing underlying units to the unconsolidated deposits of the Hanford formation (Figure 3-28). The total depth and extent of erosion in the Gable Gap area and the lithologic units particularly interbeds of the Ellensburg Formation, which have been exposed to overlying materials, are uncertain. The Rattlesnake Ridge, and possibly the Selah interbeds, may be exposed to overlying unconsolidated sediments in the Gable Gap area.

In two boreholes north of the 200 East Area (boreholes 699-53-55 and 699-55-55) approximately 2 km (1.2 mi) north of the 200-BP-1 operable unit, the first formation found beneath the Hanford formation is the Rattlesnake Ridge interbed of the Ellensburg Formation (Hoffman et al. 1992) indicating the presence of a "erosional window" through the Elephant Mountain member. Figures 3-21 and 3-22 show the inferred location of the erosional window and its relationship to overlying Hanford sediments. Although this "erosional window" is portrayed in Figure 3-29 as an oval hole of limited extent, the true size, shape and orientation of the feature is uncertain due to limited data. The "erosional window" may be continuous with the Gable Gap erosional feature. Based on the direction of flood waters, the orientation of the erosional window would be NW-SE. The results of an analysis of groundwater flow in the erosional feature, which are presented below in Section 3.6.2.4 (Vertical gradients/Aquifer communication), suggest that the feature is not an isolated "erosional window" but is likely continuous with the Gable Gap erosional feature.

North of the northeastern corner of the 200 East Area, the Elephant Mountain Member thins and an additional erosional window may be present (Last et al. 1989). The presence of this erosional window is inferred from indirect evidence, as discussed in Section 3.6.2 (Local Hydrogeology).

3.4.3.2.4 Ringold Formation. The Ringold Formation is present in the southern portion only of the study area where it unconformably overlies basalt (Lindsey et al. 1992).

The Ringold Formation is absent from the north-central part of the area, including the area immediately beneath the 200-BP-1 operable unit and to the north where sediments of the overlying Hanford formation directly overlie basalt or sedimentary interbeds in the basalt. A structure contour map depicting the top of the Ringold Formation throughout the study area is shown in Figure 3-30.

3.4.3.2.5 Plio-Pleistocene and Early "Palouse" Soil. The Plio-Pleistocene and Early "Palouse" Soil are not found in or near the 200 East Area. They are only found in the extreme western part of the study area near the eastern boundary of the 200 West Area. These units are described in detail in Last et al. (1989).

3.4.3.2.6 Hanford formation. The Hanford formation, consisting of sediments deposited during cataclysmic flooding, is continuous over all of the study area except for Gable Mountain where basalt is exposed. Generally, it overlies basalt where the Ringold Formation is not present and overlies Ringold Formation elsewhere. A geologic contour map of the Pre-Hanford formation depositional structure for the study area is shown in Figure 3-30. The Hanford Formation was reported in logs for boreholes drilled during this investigation. The total thickness of the Hanford formation throughout the study area is shown in Figure 3-31.

As discussed in Lindsey et al. (1992), the Hanford formation deposits are divided into three facies: the gravel-dominated, sand-dominated, and silt-dominated. In the vicinity of the 200 East Area, the sequences comprising the Hanford formation consist mostly of the gravel-dominated and sand-dominated facies. The gravel-dominated facies generally consist of coarse-grained basaltic sand and granule to boulder gravel. Other clast types include Ringold and Plio-Pleistocene rip-ups, granite, quartzite and gneiss (Hoffman et al. 1992). Silt-dominated facies are relatively rare except in areas south of the 200-BP-1 operable unit. Sediments typical of the gravel-dominated facies comprise most of the lower and upper gravel sequences. Thin (<1 m[<3.0 ft]) silt- and sand-dominated facies probably are intercalated throughout these gravel sequences. The third sequence, where it occurs, consists of the sand-dominated facies with lesser intercalated occurrences from both the gravel-dominated and silt-dominated facies. This sandy sequence is generally situated between the lower and upper gravel sequences. A series of cross-sections and a fence diagram depicting the relationships of these units are included in Figures 3-20 to 3-25.

Based on the distribution of these three facies in the 200 East Area and at the 200-BP-1 operable unit, the Hanford formation is divided locally into three primary stratigraphic sequences (Connelly et al. 1992; Lindsey et al. 1992; Last et al. 1989; Hoffman et al. 1992): (1) the lower gravel (Hlg), (2) the sandy unit (Hs), and (3) the upper gravel (Hug). Because of variability of Hanford formation sediments, contacts between these sediments can be difficult to distinguish, especially where the sandy sequence is missing and the upper gravel directly overlies the lower gravel. Where it is not possible to differentiate, the Hanford formation is labeled undifferentiated (Hun). Locally, discontinuous silt lenses are found throughout the gravel and sand bodies (Hoffman et al. 1992). Figure 3-20 shows the relationships of these units. Table 3-4 is a table of elevations and thicknesses of Hanford formation sequences in borings located in the vicinity of the 200-BP-1 operable unit.

The lower gravel sequence consists largely of gravel-rich strata typical of the gravel-dominated facies. Discontinuous intervals dominated by the sand-dominated facies and localized horizons of silt-dominated deposits also are encountered. Beneath the operable

unit, the lower gravel sequence ranges in thickness from approximately 5.5 m (18 ft) at 299-E33-41 to 21.9 m (72 ft.) at 299-E33-30 (Table 3-4). The average thickness of the unit is about 15 m (49 ft). The lower gravel thickens to the north of the operable unit. At boreholes 699-48-50 and 699-52-52 fine strata are absent and the lower gravels interfinger with other Hanford gravel deposits (Figures 3-22 and 3-23). Where this occurs stratigraphic sequences are not defined and the Hanford formation is undifferentiated. The lower gravel also thickens into the erosional window defined by wells 699-53-55 and 699-55-55 (Figure 3-21). Presumably this thickening is associated with the position of the main floodways.

The middle sand sequence is dominated by deposits of the sand-dominated facies. Intercalated horizons typical of both the gravel-dominated and sand-dominated facies also occur within the sequence. The middle sand sequence ranges in thickness from 39.6 m (130 ft) at 299-E33-40 to 59.4 m (195 ft) at 299-E33-41 thick beneath the operable unit. The average thickness of the unit is 51.3 m (168 ft). North and east of the operable unit (at boreholes 699-48-50 and 699-52-52), the middle sand sequence thins and eventually is absent where the Hanford formation is undifferentiated (Figures 3-21 to 3-24).

Deposits comprising the upper gravel sequence are typical of the gravel-dominated facies. Gravels in the sequence tend to be more cobble-rich in the vicinity of the flood channels located north of the operable unit than in the immediate vicinity of the operable unit. Lenticular horizons of sand-dominated and silt-dominated facies are common in the upper gravel sequence. The thickness of the upper gravel ranges from 2.4 to 16.8 m (8 to 55 ft) directly beneath the operable unit. North of the operable unit the sequence thickens considerably with the maximum thickness occurring in the erosional window at borehole 699-55-55 (55.5 m [182 ft]). As with the other two sequences, the upper gravel is absent in the vicinity of boreholes 699-48-50 and 699-52-52 where the Hanford formation is undifferentiated (Figures 3-22 and 3-23).

Compared to the surrounding area, the upper gravel appears to be relatively thin at boreholes 299-E33-12, 34, 38 and 39. An explanation for this may be construction activity around the operable unit associated with the tank farms, cribs and RCRA burial grounds.

Recent drilling in the operable unit and 200 East Area revealed the presence of several relatively continuous silt layers (Hoffman et al. 1992). These silt horizons tend to be consolidated, range from 0.3 m (1 ft) to over 1.5 m (5 ft) thick, and locally show evidence of burrowing and rooting. These properties differ from silts found interstratified within the gravely and sandy deposits that dominate the Hanford formation in the area. Large clasts also are occasionally noted in these horizons during drilling. Figures 3-21 through 3-24 show these silt horizons extending from borehole 299-E33-40 through boreholes 699-49-55B, 699-49-57B, 699-52-54. and 699-52-57. No silt horizons were found in boreholes 699-50-53, 699-48-50 and 699-52-52.

The most significant of these silt layers is a zone which is encountered at a depth of approximately 58 m (190 ft) beneath the 200-BP-1 operable unit and in the area immediately to the north. As seen in Figure 3-21 and 3-25, the horizon is relatively continuous beneath the crib area. At boreholes 299-E33-38 and 299-E33-40, which are respectively located just south and north of the crib area, the top of the horizon was found at a depth of 57.6 and 58.2 m (189 to 191 ft), respectively, and was about 1.5 m (5 ft) thick. At vadose zone boreholes 299-E33-296, -302, and -304, the zone was encountered at intervals of 57.9 to 58.4 m (190 to 191.5 ft), 58.1 to 58.2 m (190.5 to 191 ft) and 60.1 to 60.4 (197.3 to 198 ft), respectively.

Because of the presence of pedogenic alteration (burrows, root casts, calcium carbonate), it is presumed that the silt horizons are paleosols which formed as soils between flood events. The significance of these layers lies in their potential to act as perching layers for infiltrating water. Perching, if it were to occur, could lead to lateral spreading of chemical and radiological contaminants from crib operations or tank leaks. Root casts could act as a conduit which could permit infiltrating water to readily pass through the silt horizons without perching the infiltrating water.

3.4.3.2.7 Holocene Surficial Deposits. Holocene surficial deposits consist of silt, sand and gravel that form a (<10 m [33 ft]) veneer across much of the Hanford site. These sediments were deposited by a combination of eolian and alluvial processes. Eolian activity in the vicinity of the 200-BP-1 operable unit has done little but to locally rework and redistribute surficial deposits. This has produced sheet sands which blanket flood topography.

3.4.3.3 Local Geologic Structure. The study area is situated between the axes of the Gable Mountain anticline and Cold Creek syncline, on the south flank of the Gable Mountain anticline. The Gable Mountain anticline dips at about 2 degrees overall, but locally dips as much as 10 degrees in to the Cold Creek syncline (Fecht 1978). The structure of this area is characterized by a complex series of doubly plunging en echelon anticlines and synclines. These structures are interpreted as parasitic folds situated within the closure of the larger major anticlinal fold; the eastern extension of the Umtanum Ridge. The main axis of the anticline lies about 3 to 4 km north-northeast of the 200 East Area. Parasitic fold axes occurring within the study area trend southeast to northwest en echelon. The axial trends of individual folds are generally curvilinear and are either doubly plunging or subdued by surrounding folds with higher amplitudes and greater wavelengths (Fecht 1978).

Basalt bedrock beneath the operable unit dips to the south (Figure 3-29). North of the operable unit two distinct anticlinal features are found in the basalt. These features are depicted in Figure 3-30. One of these subsidiary anticlines lies about 1 to 2 km (0.6 to 1.2 mi) northeast of the 200 East Area and the other lies on the northern boundary of the 200 East Area. Both subsidiary anticlines trend northwest-southeast. South of the two subsidiary anticlines, basalt flows, sedimentary interbeds, and Ringold Formation dip southward into the Cold Creek syncline.

Cataclysmic flooding in this area is inferred to have been preferentially directed through the synclinal areas scouring and enlarging the geometry of structural features. The erosional window north of the operable unit was produced by downcutting in these structural areas (Hoffman et al. 1992).

In Figure 3-30, both of the subsidiary anticlines located immediately north of the 200 East Area are depicted above the current water table (about 123 m elevation). Figure 3-30 is adapted from work presented in Lindsey et al. (1992).. Another interpretation (Figure 3-29) is presented in Connelly et al. (1992) , however, where it is suggested that only the northernmost of the two subcrops actually occurs. In addition, the shape of the northernmost subcrop differ significantly in the two interpretations. It should be emphasized, therefore that there is uncertainty with respect to the dimensions and areal extent of these basalt subcrop features. The cause of the differing interpretations may be related to changes in the water table elevation over time.

3.5 PEDOLOGICAL CHARACTERISTICS

The term "pedology" is used to refer broadly to the study of the nature, properties, formation, distribution, classification, function and use of soils. The term "soil" is also used broadly as a synonym for regolith, or all unconsolidated materials which overlie bedrock.

Pertinent soil characteristics provided in this section include soil classification, and general engineering and physical properties for the regional and local scales.

3.5.1 Regional Soil Characteristics

The earliest study of soils in Benton County, which includes most of the Hanford Site, was performed in 1916 by Kocher et al. (1921). Maps generated from this survey indicate that the soils in the Hanford Site belong within four major groups that can be classified according to their origin. The four groups included:

- Soils derived from loessial or wind-blown material
- Soils derived from eolian or wind-blown material
- Soils derived from old valley-filling material, mainly lake-laid
- Soils derived from stream laid material.

Kocher et al. (1921) mapped 26 classes of soils within these four groups, and three classes of miscellaneous nonagricultural material, including scabland, river wash, and dune sand.

The major soil types mapped in the north-central portion of the Hanford Site included Ephrata sandy loam, Ephrata fine sandy loam, Ephrata sand, Winchester sand, and Winchester fine sand. The soils of the Winchester series are basaltic, loose and open, and of low moisture-retaining capacity. The Ephrata series overly a substratum of gravels and sand and are somewhat calcareous. The gravely phase and the heavy phase of the Ephrata fine sandy loam are reported by Kocher et al. (1921) to be the soils of greatest agricultural significance in Benton County.

In a later study (Western States Land Grant Universities and Colleges and Soil Conservation Service [SCS] 1964), which consisted of a generalized soil survey of the western United States, the soils of the Hanford Site area were characterized as largely immature soils formed on unconsolidated upland materials and eolian sands with few clearly-defined horizons.

Few, or no, clearly defined soil horizons are present in regosols, or soils largely dominated by the characteristics of the parent materials. The regosols of the Hanford Site occur on glaciofluvial deposits that have been continually shifted and sorted by wind-erosion and deposition. These soils support a shrub-steppe vegetation community, and are principally used for grazing and limited irrigation crop production (SCS 1960).

Hajek (1966) presents a soil map and descriptive report of soils in the Benton County portion of the Hanford Site. On the basis of morphologic and genetic characteristics, 13 soil types were identified. An approximate land use capability classification is provided for these soils, on the basis of soil limitations for, and damage risks associated with,

agricultural use. Approximate engineering classifications for these soils, using the Unified Soil Classification System, are also provided in Hajek (1966).

3.5.2 Local Soil Characteristics

In a soil survey of the Hanford Site (Hajek 1966), three major soil types were identified in the vicinity of the 200 East Area, each of which is approximately equally represented in areal extent. These include the Ephrata Sandy Loam, the Burbank Loamy Sand, and the Rupert Sand. The Ephrata Sandy Loam and Burbank Loamy Sand appear to be present within the operable unit, although recent field studies were not conducted for confirmation. Surface soils over crib areas are disturbed and represent back-filled materials that are not representative of native surface soils. Each of these soils is discussed below. Measured physical properties of the operable unit soils are discussed. The soil types mapped in the Benton County portion of the Hanford Site are shown in Figure 3-32.

3.5.2.1 Soil Types

3.5.2.1.1 Ephrata Sandy Loam. Occurring to an average depth of 30 cm (12 in.), the Ephrata sandy loam is a dark grayish brown, medium-textured soil underlain by deep gravely material. The topography is generally level. The surface of the Ephrata sandy loam belongs to Group SM (silty sand) to ML (silt), and the subsurface belongs to Group ML (silt). Group ML (silt) are fine-grained soils composed of silts and clays with little or no plasticity.

3.5.2.1.2 Burbank Loamy Sand. The Burbank loamy sand is a dark grayish brown, coarse-textured, excessively-drained soil underlain by gravel. The surface soil is usually about 40 cm (16 in.) thick but can be as much as 75 cm (30 in.). The gravel content of the subsoil may range from 20 to 80% (by volume). The surface of the Burbank loamy sand is Group SM (silty sand) and the subsoil is group GM (silty gravel) to GP (poorly-graded gravel). Group GM (silty gravel) are coarse-grained soils composed predominantly of gravels with more than 12% fines. Group GP (poorly-graded gravel) contains coarse-grained soils that are predominantly well-sorted gravels with less than 5% fines.

3.5.2.1.3 Rupert Sand. The Rupert sand represents one of the most extensive soils at the Hanford Site. The soil is a brown to grayish brown, moderately-deep, coarse sand. Rupert soils developed under grass and sagebrush in coarse alluvial deposits mantled by wind-blown sand. Relief characteristically consists of hummocky terraces and dune-like ridges. The surface and subsoil of the Rupert sand were assigned to Group SM (silty sand) which consists of coarse-grained soils composed predominantly of sands with more than 12% fines.

3.5.2.2 Soil Physical Properties. Surface soil and subsurface soil samples were collected for physical analyses under Tasks 2, 4, and 6 of the 200-BP-1 Operable Unit Phase I RI field investigation. Samples were tested for moisture content, porosity, saturated and unsaturated hydraulic conductivity, specific gravity, dry bulk density, calcium carbonate content and grain size. The results of this work are summarized in Hoffman (1992) and Hoffman et al. (1992). The discussion which follows is based primarily upon these technical sources.

This section will address the physical characteristics of soils within only the upper 4.6 m (15 ft) of the 200-BP-1 operable unit. The 4.6 m (15 ft) depth designation is consistent with evaluations of risk presented in subsequent chapters. Discussion of the properties and characteristics of deeper, subsurface materials, as well as the results of all soil permeability measurements, will be presented and discussed in Section 3.6 (Hydrogeological Characteristics). Results referred to in the discussion which follows are taken from only those borings completed within the 200-BP-1 operable unit or immediate vicinity. The locations of Task 2, 4, and 6 boreholes are indicated in Figures 2-1 and 2-5. Tables 2-2 and 2-3 summarize the drilling performed under Tasks 2, 4, and 6. The results of all physical testing completed to date are presented in Hoffman (1992). Grain-size analysis results for 200-BP-1 operable unit samples are presented in Table 3-5.

The surficial soil characteristics of the 200-BP-1 operable unit were significantly modified in the UN-200-E-89 interim stabilization effort. As shown in Figure 2-3, an approximately 4-ha (10-ac) area in the eastern half of the 200-BP-1 operable unit was scraped of the uppermost 15 cm (6 in.) of soil to reduce the level of surface soil radiation. This soil was collected, consolidated, and re-placed atop the 216-B-43 through -50 cribs and then covered with approximately 46 to 61 cm (18 to 24 in.) of clean soil. Other areas within the 200-BP-1 operable unit (Figure 2-3) were also covered with clean soil (15 to 30 cm [6 to 12 in]) to facilitate drilling during the RI activities. Areas outside the 200-BP-1 operable unit, specifically the steep embankments on the east and north sides of the BY Tank Farm were stabilized with large cobble and pit run soil (Hayward 1992). Due to this disturbance, as well as additional surface disturbances which have likely occurred in the past due to general construction activities, the in-place surficial soils probably do not have the same properties as the undisturbed major soil types. The Task 2, 4, and 6 drilling, however, was undertaken after the UN-200-E-89 interim stabilization work. The samples collected and tests performed are therefore representative of the current, in-place soil conditions.

3.5.2.2.1 Grain-Size Distribution in Surface Soils. Grain-size distribution analyses were completed on 17 near surface samples (less than 4.6 m [15 ft] in depth) within the 200-BP-1 operable unit. The samples were collected from monitoring wells 299-E33-38, -39, and -40, as well as from six Task 2 and 4 borings drilled through the cribs. Four of the samples were collected from within the infiltration gravels, based on estimated depths of the top and bottom of the cribs observed during drilling (Table 2-2). The results of these analyses are presented in Table 3-5. The soil fraction exceeding 0.84 mm (0.033 in.) is provided to allow for subsequent wind erosion estimates in Chapter 5.0 (Contaminant Fate and Transport Analysis) in accordance with Skidmore and Woodruff (1969).

Grain size analyses indicate that the surface soils are predominantly gravels with some sands. Fines contents (smaller than 0.074 mm [0.0029 in.]) ranged from 4 to 24% and averaged approximately 10%. Sands (between 2 mm and 0.074 mm [0.0787 and 0.0029 in.]) ranged from 4 to 78% and averaged approximately 30%. Gravel contents ranged from 14 to 90% and averaged 60%. The soil fraction exceeding 0.84 mm (0.033 in.) ranged from 8% to 79% and averaged approximately 29%.

No definite grain size trend with depth is apparent (Table 3-5). However, it appears that the shallower soils tend to be characterized by slightly higher fines contents than the deeper surface soils. The average soil fraction exceeding 0.84 mm (0.033 in.) of soils collected at depths of 1.5 m (5 ft) or less was 40%.

Four samples were collected in the crib infiltration gravels (Table 3-5). These were collected at cribs 216-BY-57 and -61 at depths of from 3.0 to 4.6 m (9.8 to 15.1 ft). The fines and sands contents of these gravels is low, averaging 8 and 11%, respectively. The average soil fraction exceeding 0.84 mm (0.033 in.) in these gravels is approximately 14%.

3.5.2.2.2 Soil Moisture Content. Moisture content was low and ranged from 1.4 to 5.57% (by weight percent). The average value was 2.6%. These values indicate the water content of the soils to be near or at residual levels.

3.6 HYDROGEOLOGICAL CHARACTERISTICS

This subsection presents the regional and local hydrogeology for the 200-BP-1 operable unit. The discussion on regional hydrogeology summarizes groundwater conditions in the Pasco Basin, detailing the primary aquifers and providing the regional context necessary to understand the local hydrogeology. The local hydrogeology discussion is focused at the study area scale and relies primarily on data presented in Connelly et al. (1992).

3.6.1 Regional Hydrogeology

The hydrogeology of the Pasco Basin has been broadly characterized as consisting of four primary hydrogeologic units (DOE 1988). These units correspond to the upper three formations of the Columbia River Basalt Group (Grande Ronde Basalt, Wanapum Basalt, and Saddle Mountains Basalt) and the sedimentary overburden. The basalt aquifers consist of the flood basalts of the Columbia River Basalt Group and relatively minor amounts of intercalated fluvial and volcaniclastic sediments of the Ellensburg Formation. Confined zones in the basalt aquifers are present in the sedimentary interbeds and/or interflow zones that occur between dense basalt flows. The main water-bearing portions of the interflow zones are networks of interconnecting vesicles and fractures of the basalt flow tops and flow bottoms (DOE 1988). The suprabasalt sediment, or uppermost aquifer system, consists of fluvial, lacustrine, and glaciofluvial sediments. This aquifer is regionally unconfined and is contained largely within the Ringold Formation and Hanford formation.

The uppermost aquifer is part of a flow system that is local to the Pasco Basin, as are the uppermost basalt interbed aquifers (Gephart et al. 1979; DOE 1988). Groundwater in these aquifer systems is probably recharged and discharged locally. Deeper in the basalt, interbed aquifer systems are part of the regional, or interbasin, flow system, which extends outside the margins of the Pasco Basin (DOE 1988). Groundwater in the uppermost aquifer system is regionally unconfined and occurs within the glaciofluvial sands and gravels of the Hanford formation and the fluvial/lacustrine sediments of the Ringold Formation. Confined to semi-confined aquifers of more limited extent also occur in the suprabasalt sediments of the Pasco Basin. These confined zones are generally located within the local flow system, between the unconfined aquifer and the underlying basalt surface. Further discussion of the aquifer system is provided below.

3.6.1.1 Unconfined Aquifer. The unconfined aquifer is laterally extensive, occurring below most of the Hanford Site with saturated thicknesses ranging up to 90 m (295 ft) under the 200 West Area. The unit thins and is locally absent along the flanks of anticlinal structures (i.e., Gable Mountain/Gable Butte and Yakima Ridge) (Gephart et al. 1979). The base of the

unconfined aquifer is generally defined as the top of the uppermost basalt flow. Fine-grained overbank and lacustrine deposits of the Ringold Formation, however, locally form confining or semi-confining layers for underlying Ringold fluvial gravels.

The main body of the unconfined aquifer generally occurs within the sediments of the Ringold Formation. In the southwestern portion of the Pasco Basin, the position of the water table is generally within Ringold fluvial gravels. In the northern and eastern Pasco Basin, the water table generally occurs within the Hanford formation.

3.6.1.1.1 Recharge. Natural recharge to the unconfined aquifer occurs primarily from run-off of precipitation from higher elevation areas including Saddle Mountains, Umtanum and Yakima ridges, and Rattlesnake Hills (Deju and Fecht 1979; Gephart et al. 1979; DOE 1988), as well as water infiltrating from small ephemeral streams. The Yakima and Columbia rivers also contribute to the natural recharge, as may the deep basalt aquifers (DOE 1988).

The movement of precipitation through the unsaturated (vadose) zone has been studied at several locations on the Hanford Site (Isaacson et al. 1974; Jones 1978; Gee and Heller 1985; Gee 1987; Routson and Johnson 1990; Rockhold et al. 1990). Conclusions from these studies vary, however, the estimates of deep percolation to the uppermost aquifer are consistently low (from 0 to 7.87 cm/yr [0 to 3.1 in/yr]). Little, if any, recharge to the groundwater occurs from percolating rainfall on the broad areas of the desert terrain because of the high rates of evapotranspiration. Gee (1987) and Routson and Johnson (1990) concluded that no downward percolation of precipitation occurs on the 200 Areas Plateau where the sediments are layered and vary in texture, and that all moisture penetrating the soil is removed by evapotranspiration.

The most important source of recharge to, and water level change within, the unconfined aquifer on the Hanford Site is from onsite waste-water disposal. Recharge from waste-water disposal was estimated to be about 5.5×10^7 l/d (1.4×10^7 gl/d) or about 10 times the amount of natural recharge entering the unconfined aquifer system within the Cold Creek Valley (DOE 1988).

Artificial recharge of the unconfined aquifer system occurs from the disposal of large volumes of wastewater on the Hanford Site and from large irrigation projects surrounding the Hanford Site. Recharge through ponds and cribs in the 200 Areas is the largest single artificial recharge source, beginning in the late 1940s and continuing to the present. Other artificial recharge sources include irrigation loss west of the 200 Areas (Graham 1983), infiltration ponds at Advanced Nuclear Fuels Corp (USGS 1978), and infiltration ponds at the City of Richland well field (CWC-HDR, Inc. 1988).

3.6.1.1.2 Movement. Figures 3-33 and 3-34 illustrate the groundwater table for the Hanford Site during the periods January 1944 and June 1989, respectively. As seen in the figures, effluent disposal has altered the groundwater flow directions and gradients at the Hanford Site. Before operations at the Hanford Site began in 1944, the hydraulic gradient in all but the southwestern-most portion of the Hanford Site was approximately 0.9 m/km (5 ft/mi). Regional groundwater flow was generally toward the east-northeast, although flow north of Gable Mountain was more to the north. Groundwater flow north of Gable Mountain now trends in a more northeasterly direction as a result of mounding near reactors and flow through Gable Gap. South of Gable Mountain, flow is interrupted locally by the groundwater mounds in the 200 Areas. Under the influence of mounding,

groundwater flow in the 200 East Area is radial with portions heading northward, passing between Gable Mountain and Gable Butte (Delaney et al. 1991).

Over the period 1950 to 1980, water levels in the unconfined aquifer are reported to have risen by as much as 3.7 m (12 ft) in the 200 East Area and 24 m (80 ft) in the 200 West Area (DOE 1988). The rate of increase was most rapid from 1950 to 1960; the rate of increase was slower from 1960 to 1970. From 1970 to 1980, only small increases in water table elevation occurred, and the unconfined aquifer appears to have been in approximate steady-state with recharge sources. This rise in water-table elevations increased the potential for downward movement of groundwater from the unconfined to the confined basalt and interbed aquifers. The degree of exchange which occurred between the groundwater systems is not known. Aquifer communication is discussed further in Section 3.6.2.4 (Vertical Gradients/Aquifer Communication).

Studies have shown that the existing general flow pattern may reverse and return to the pre-operational pattern if the artificial recharge were discontinued, allowing the groundwater mound to dissipate (DOE-RL 1990a). Data presented in Kasza et al. (1992) indicate that this expected mound dissipation is occurring in the 200 Areas. Water level data from 1988 most nearly corresponds to the highest groundwater levels measured in the recent past. A general lowering of the water table is occurring beneath the 200 Areas in response to the closure of the Gable Mountain pond and the U pond, and the decrease in disposal of process water to B pond. From December 1988 to December 1991, the water table beneath the 200 Areas decreased in elevation by as much as 1 m (3.5 ft). To the north of the 200 East Area, in the vicinity of West Lake, the decrease was lower (about 0.5 m [1.5 ft]). The change in water table elevation in the 200 Areas over this period is depicted in Figure 3-35.

3.6.1.1.3 Discharge. Groundwater discharge from the unconfined aquifer is almost exclusively to the Columbia River along the eastern and northeastern margins of the Pasco Basin (Deju and Fecht 1979; Gephart et al. 1979; DOE 1988). Downward leakage to the lower confined aquifers may be occurring under the eastern groundwater mound beneath B Pond and through features such as erosional windows discussed in Section 3.4.3 (Local Geology). Communication between the uppermost aquifer and the underlying Rattlesnake Ridge aquifer will be discussed in Section 3.6.2 (Local Hydrogeology).

West Lake is hydraulically connected to the unconfined aquifer and represents a topographic depression that intersects the water table. Because of high water evaporation rates and low surface overland flow, the lake is expected to result in a net loss of groundwater, and thus constitute a local discharge zone (DOE-RL 1990a).

3.6.1.1.4 Hydraulic Properties. Hydraulic conductivity estimates for the unconfined aquifer have been mapped over the Hanford Site, as shown in Figure 3-36 (DOE 1988). The hydraulic conductivities were obtained from pumping tests (Biershenk 1957; Kipp and Mudd 1973) and are not layer specific, but apply to the combined conductivity of all layers stressed during the test. The hydraulic conductivity range is from approximately 10^{-3} to 10^{0} cm/s (10^{0} to 10^{3} ft/d), reflecting heterogeneity of the soils. Transmissivities vary widely regionally because of the variable saturated thickness of the unconfined aquifer.

Generally, saturated hydraulic conductivity is greater in the Hanford formation, where values from 10^{-1} to 10^{1} cm/s (10^{2} to 10^{4} ft/d) are typical, than in the Ringold Formation where hydraulic conductivities are generally from about 10^{-5} to 10^{-1} cm/s (10^{-2} to

10^2 ft/d). Table 3-6 summarizes the anticipated ranges of saturated hydraulic conductivities for each stratigraphic interval of the unconfined aquifer.

Fewer data are available on aquifer specific yield for the unconfined aquifer. Storage coefficients determined in multiple well pumping tests from the unconfined aquifer ranged from 0.0002 to 0.2 (DOE 1988). Values determined at Hanford formation wells ranged from 0.03 to 0.2, whereas values in Ringold Formation wells were generally less than 0.06.

3.6.1.2 Confined Aquifers. Confined aquifers occur within the lower portion of the Ringold Formation, but are generally more limited in areal extent than the unconfined aquifer. In the western portion of the Pasco Basin, a confined-to-semi-confined aquifer is present within the basal unit of the Ringold Formation (as defined by DOE 1988). A thick silt deposit (the lower unit of the Ringold Formation as defined in DOE 1988) forms the aquitard between the unconfined and confined zones. Other confined-to-semi-confined zones occur locally within the middle and lower units of the Ringold Formation as a result of interfingering silt aquitards and more permeable lenses of sand and gravel. These zones appear to be laterally discontinuous and likely merge with the unconfined system.

A multiple confined aquifer system occurs within the Columbia River Basalt Group underlying the Pasco Basin (Deju and Fecht 1979; Gephart et al. 1979; DOE 1988). The confined aquifers consist primarily of interbeds within the basalt (DOE 1988). The interbeds occur between basalt flow tops of the older flows and basalt flow bottoms of the younger flows (Graham 1983). Flow interiors, comprised primarily of dense basalts, separate the interbeds forming confining aquitards.

The uppermost interbed aquifers are found in the Saddle Mountains Basalt and include, from youngest to oldest, the Rattlesnake Ridge, Selah, Cold Creek and Mabton interbeds (Figure 3-17). Interbed aquifers of the Saddle Mountains Basalt range in thickness from 6 to 35 m (20 to 110 ft) and are likely localized to the Pasco Basin by geologic structures along the basin margin (Gephart et al. 1979; DOE 1988). Deeper interbeds which occur in the underlying Wanapum and Grande Ronde Basalt formations, appear to be hydraulically connected with the regional flow system outside the Pasco Basin (DOE 1988).

3.6.1.2.1 Recharge. Recharge to the interbeds of the Saddle Mountains Basalt is obtained directly from precipitation onto the exposed basalt ridges surrounding and within the Pasco Basin (Deju and Fecht 1979; Gephart et al. 1979; DOE 1988). Leakage from the unconfined aquifer also recharges at least the uppermost interbed aquifer (the Rattlesnake Ridge interbed, which underlies the Elephant Mountain basalt member) below the 200 Areas plateau, especially where artificial recharge has caused mounding in the unconfined aquifer (Graham 1983; DOE 1988; Delaney et al. 1991; Connelly et al. 1992). In this area, erosion of the Elephant Mountain member may have lead to an enhanced hydraulic connection between the Rattlesnake Ridge interbed and the unconfined aquifer (Graham 1983).

The deeper basalt interbed aquifers, between and within the Wanapum and Grande Ronde Basalt Formations, obtain recharge waters in the Pasco Basin from vertical leakage of overlying interbed aquifers within the Saddle Mountains Basalt, and horizontal inflow from the regional flow system to the east and west.

3.6.1.2.2 Movement. Within the Pasco Basin, groundwater potentials of Saddle Mountains Basalt indicates that groundwater flow is generally from topographically high to topographically low regions (Figure 3-37), similar to flow in the unconfined aquifer (DOE 1988). Steep groundwater gradients occur on the flanks of the major anticlines, including the Horse Heaven Hills, Frenchman Hills, Rattlesnake Hills, and Saddle mountains. Lateral groundwater flow in the Saddle Mountains Basalt appears to mirror the surface topography and is generally toward major surface drainage features. The predominant generalized flow direction, therefore, across the Hanford Site is from west to east (DOE 1988).

Groundwater flow in the Wanapum and Grande Ronde basalts is thought to be controlled less by local surface drainage patterns and more by the major rivers, streams, and coulees. Potentiometric levels in the deeper interbeds of the Wanapum and Grande Ronde basalts are interpreted to have a smoother form as a consequence of being less influenced by smaller surface drainage features (DOE 1988).

3.6.1.2.3 Discharge. Potentiometric and hydrochemical data presented in DOE (1988) portray the Pasco Basin, in relation to the surrounding Columbia Plateau, as an area of regional groundwater flow convergence and probably of groundwater discharge. Regional discharge from basalts appears to take place in the topographically low and well-dissected regions of the plateau where groundwater flows into stream courses (DOE 1988).

Within the Pasco Basin, the Saddle Mountains Basalt apparently discharges along the Columbia River from the confluence of the Columbia River with the Walla Walla northward, except across the northern portion of the Hanford Site. The Saddle Mountains Basalt potentiometric surface indicates that the Columbia River is the ultimate discharge for groundwater from these Basalts in most places where it flows over the unit. The Saddle Mountains Basalt may also discharge into the lower Snake and Yakima rivers. In much of the area of discharge, the Saddle Mountains Basalt discharges to the surface through the suprabasalt sediments (DOE 1988).

3.6.1.2.4 Hydraulic Properties. Hydraulic conductivities within the basalt interbeds are generally orders of magnitude lower than those observed in the unconfined aquifer. Aquifer testing in interbeds of the Saddle Mountains Basalt yielded hydraulic conductivities ranging from 10^{-4} to 10^{-1} cm/s (10^{-1} to 10^{0} ft/d) (DOE 1988). No values of storativity are currently available. Storativity values, however, are anticipated to occur within the range commonly reported (i.e., 10^{-5} to 10^{-3}) for confined aquifers (DOE 1988).

The flow interiors of the basalt formations have hydraulic conductivities orders of magnitude lower than the interbeds, ranging from 10^{-13} to 10^{-7} cm/s (10^{-10} to 10^{-4} ft/d) (DOE 1988). Storativity estimates for the basalts have not been made, but likely range from 10^{-5} to 10^{-3} (DOE 1988).

3.6.2 Local Hydrogeology

The hydrogeologic system underlying the 200-BP-1 operable unit is generally congruent with the regional hydrogeologic model of the Hanford Site. The vadose zone consists predominantly of sandy gravel, gravely sand, and silty sandy gravel of the Hanford formation. The unconfined aquifer occurs within both the Hanford and Ringold

Formations and is continuous within the regionally extensive unconfined aquifer observed below most of the Hanford Site.

Immediately beneath the 200-BP-1 operable unit and to the north, as discussed in Section 3.4.3 (Local Geology), the Ringold Formation has been eroded by Pleistocene cataclysmic flood events. The unconfined aquifer which is beneath the 200-BP-1 operable unit extends downward through the Hanford formation to the basalt bedrock of the Elephant Mountain Basalt. The basalt separates the unconfined aquifer from the confined aquifer of the Rattlesnake Ridge interbed below. North of the 200-BP-1 operable unit, in the area between Gable Mountain and Gable Butte, the Elephant Mountain and Pomona Basalts have been eroded as well (Figures 3-26 through 3-28), and the Rattlesnake Ridge aquifer, as well as deeper interbeds, may be exposed to the overlying unconsolidated sediments.

The discussion which follows is based upon previous interpretations of the 200-BP-1 operable unit hydrogeology (DOE-RL 1990a), as supplemented by additional data which was collected in the 200-BP-1 Phase I RI field investigation. Much of this recent work was summarized and compiled into a comprehensive hydrogeologic model of the 200 East Groundwater Aggregate Area (Connelly et al. 1992). Stratigraphic conditions of the 200-BP-1 operable unit were discussed in Hoffman et al. (1992). Boring logs and the results of physical testing on surface and subsurface samples were presented in Hoffman (1992). The results of hydraulic tests performed recently on wells in the vicinity of the 200-BP-1 operable unit are described in (Swanson 1992). Delaney et al. (1991) and Lindsey et al. (1992) provide descriptions of regional and 200 East Area geology, respectively.

3.6.2.1 Vadose Zone. The vadose zone is the region above the water table in which the fluid pressures of the sediments are negative with respect to local atmospheric pressure. It occurs between the ground surface and the water table, and is the zone through which natural and man-made recharge waters may flow to the water table. This section will address vadose zone sediments which occur at depths greater than 4.6 m (15 ft). Discussion of sediments from the ground surface to 4.6 m (15 ft) is included in Section 3.5 (Pedological Characteristics).

In the vicinity of the 200 East Area, the vadose zone encompasses units of the Hanford formation and the Ringold Formation (Lindsey et al. 1992). A generalized description of these units is given in Section 3.4.2 (Regional Geology). Beneath the 200-BP-1 operable unit, however, the vadose zone consists of the interlayered sandy gravel, gravely sand, and silty sandy gravel of the Hanford formation only, as shown in the geologic cross-sections and fence diagram (Figures 3-20 to 3-25). The Ringold Formation, because of erosion which occurred during cataclysmic flooding, has been removed from the immediate vicinity of the 200-BP-1 operable unit and the areas to the north. The areal extent of the Ringold Formation is depicted in Figure 3-30.

3.6.2.1.1 Stratigraphy and Thickness. The total thickness of the vadose zone in the study area varies from approximately 75 m (245 ft) directly beneath the 200-BP-1 operable unit to between 25 to 50 m (82 to 165 ft) in the Gable Mountain/Gable Butte area. In the study area, except for localized areas where basalt bedrock occurs above the water table, all portions of the vadose zone are comprised of the Hanford formation.

The importance of the vadose zone in the 200-BP-1 operable unit Phase I RI study lies principally in how it has affected the distribution of contaminants contained in waste

waters disposed of in the 216-B cribs. As such, the description which follows will be focused primarily on the materials which directly underlie the 200-BP-1 operable unit. The properties of the vadose zone in other portions of the study area are largely irrelevant to this study.

Beneath the 200-BP-1 operable unit, the vadose zone is comprised of three primary lithologic units: the Hanford upper gravel, the Hanford sand, and the Hanford lower gravel. The water table occurs within sediments of the lower gravel. Figure 3-25 depicts the stratigraphic relationships between these three units.

Based on information collected from two boreholes drilled in the immediate vicinity of the cribs (299-E33-40 and -38), the lower gravel unit is approximately 14 to 20 m (45 to 65 ft) in thickness directly beneath the 216-B cribs (Figure 3-21). The sand unit is the thickest of the three lithologies and varies from approximately 30 to 41 m (100 to 135 ft) in thickness while the upper gravel sequence ranges from approximately 5 to 14 m (15 to 45 ft) in thickness beneath the cribs. The upper gravel and the sand generally thicken from south to north beneath the crib area while the lower gravel thins.

In addition to the three main stratigraphic units of the Hanford Formation, several relatively continuous silt layers were revealed during the recent drilling (Hoffman et al. 1992). The most significant of these is a thin silt horizon identified at a depth of approximately 58 m (190 ft) which appears to be relatively continuous beneath the 200-BP-1 operable unit and crib area. As seen in Figure 3-21 through 3-25, the horizon is found beneath the cribs and extends north through 699-49-55B, 699-49-57B, and 699-52-54. Beneath the 200-BP-1 operable unit, the layer is continuous between boreholes 299-E33-40, 299-E33-38, 299-E33-31, 299-E33-32, 299-E33-41, 299-E33-34, and 299-E33-30 (Figure 3-25). The unit was also detected in the three Task 4 (deep vadose zone drilling) boreholes 299-E33-296, 299-E33-302 and 299-E33-304 which were drilled directly through cribs 216-B-43, 216-B-49, and 216-B-57, respectively. The unit was apparently not detected in boreholes 299-E33-12 and 299-E33-39, located directly east of the 200-BP-1 operable unit (Figure 3-25).

The silt horizon at a depth of about 58 m (190 ft) is the only fine-grained layer observed beneath the 200-BP-1 operable unit which is relatively continuous beneath the entire crib area. The thickness of the zone beneath the 200-BP-1 operable unit varies from approximately 0.15 to 3.4 m (0.5 to 11 ft). Directly beneath the cribs, the zone is from about 0.15 m (0.5 ft) in thickness at borehole 299-E33-302 to 1.5 m (5 ft) at 299-E33-40. The dip of the silt horizon, based on the observed elevations in boreholes 299-E33-30, 299-E33-34, 299-E33-38 and 299-E33-40, is gentle and generally is toward the north at about 1 degree.

3.6.2.1.2 Grain Size Data. In section 3.5.2 (Local Soil Characteristics), the grain size characteristics of the surface soils were reviewed. In this section, data collected for subsurface materials (> 4.6 m [15 ft] in depth) will be presented.

As seen in Table 3-5, samples collected were predominantly sands and gravels with only a few examples of fine-grained material. The fines contents of the soils were generally low, typically less than 10%. In boreholes 299-E33-38 and -40, a fine grained layer was identified at depths of 58 and 59 m (190 and 195 ft), respectively. The fines contents of the two samples were 87 and 23%, respectively. This layer is felt to be continuous beneath the crib area, as discussed above.

3.6.2.1.3 Hydraulic Characteristics. Although a variety of methods have been developed to measure unsaturated hydraulic conductivity, these are often costly and difficult to implement. An alternative to direct measurements is the use of theoretical approaches that predict unsaturated hydraulic conductivity relationships from measured soil water retention data. As such, under Task 4 of the 200-BP-1 operable unit work plan (DOE-RL 1990a) and other 200 area RCRA and CERCLA projects, water retention measurements (for both wetting and drying) were made on 60 unsaturated zone samples collected from 13 boreholes located throughout the 200 East Area, three of which are located in 200-BP-1 operable unit (299-E33-38, -40 and -307 (Figures 2-1 and 2-5)). The water retention data were used in conjunction with the theoretical approach of van Genuchten et al. (1991) to define the unsaturated hydraulic conductivity vs soil moisture content relationship for each sample. The theoretical basis of this approach and results, as well as all retention curves and hydraulic conductivity vs matric potential curves, are summarized in Connelly et al. (1992). Detailed presentations of the work are provided in Smoot et al. (1989) and Bergeron et al. (1987).

For the Hanford gravels, a total of fifteen soil moisture retention curves were measured, including eight curves for samples from the 200-BP-1 operable unit boreholes 299-E33-307, 299-E33-38, and -40. The functional relationships, which are presented in Connelly et al. (1992) exhibit strong non-linear characteristics, i.e. for a small change in volumetric moisture content, the hydraulic conductivity change can be several orders of magnitude. The total range of unsaturated hydraulic conductivities calculated for the Hanford gravels was from approximately 10^{-2} to 10^{-20} cm/s (10^{1} to 10^{-17} ft/d). Saturated hydraulic conductivities ranged from approximately 10^{-6} to 10^{-3} cm/s (10^{-3} to 10^{0} ft/d). The low hydraulic conductivity value was measured in a sample of sandy clay collected at 57.9 m (190 ft) depth from borehole 299-E33-38. The high value was measured in a sandy gravel collected from borehole 699-42-41B. These results indicate a high degree of variability within the Hanford gravels with respect to their hydraulic properties.

A total of 41 samples were evaluated for the Hanford sand sequence of which three were collected from the 200-BP-1 operable unit (from borehole 299-E33-38). As with the gravels, the degree of non-linearity is high. The range of unsaturated hydraulic conductivities was similar to that exhibited for the gravels. Saturated hydraulic conductivities ranged from approximately 10^{-5} to 10^{-2} cm/s (10^{-2} to 10^{1} ft/d), indicating a high degree of variability as with the gravels.

The results of the unsaturated hydraulic conductivity measurements indicate that both saturated and unsaturated hydraulic conductivity can vary by several orders of magnitude within a lithologic unit. As discussed in Connelly et al. (1992), it is necessary to evaluate sample collection and textural description when selecting moisture characteristic curves. Typically, the choice of a water retention curve is made by examining water retention curves along with the particle size analyses to distinguish lithologies with significantly different soil textures and hydraulic conductivities.

Additional saturated hydraulic conductivity data is included in Hoffman (1992) where the results of physical soils testing performed under Tasks 2, 4, and 6 of the work plan are presented. Hydraulic conductivities presented include the results of vertical hydraulic conductivity measurements. Results range from 10^{-2} to 10^{-6} cm/s (10^{1} to 10^{-3} ft/d). Most of the values fell in the 10^{-3} to 10^{-5} cm/s (10^{0} to 10^{-1} ft/d) range.

3.6.2.2 Uppermost Aquifer. This section will describe the local hydrogeologic characteristics of the uppermost aquifer.

3.6.2.2.1 Occurrence and Thickness. The uppermost aquifer system within the study area consists of the saturated units above the uppermost basalt surface as defined by Delaney et al. (1991) and Connelly et al. (1992). The aquifer system generally includes the Hanford formation and Ringold Formation. The Ringold Formation, while present in the study area vicinity, does not occur immediately beneath the 200-BP-1 operable unit or in areas to the north, and therefore is not a primary hydrostratigraphic unit of interest in this report.

The base of the uppermost aquifer system is typically the Elephant Mountains Basalt member of the Saddle Mountains Basalt Formation and/or the Umatilla/Pomona Basalt. Local areas exist where the Elephant Mountain Member has been removed by erosion during cataclysmic flooding (Lindsey et al. 1992; Hoffman et al. 1992). Such erosion has occurred north of the 200-BP-1 operable unit in the vicinity of wells 699-53-55 and 699-55-55, and in the Gable Gap area (Figure 3-28). Here, stratigraphically lower basalts are the base of the uppermost aquifer system. Figure 3-29 is a structure contour map of the top of the uppermost basalt in the study area.

The relatively more permeable units of the uppermost aquifer system include the sands and gravels of the Hanford formation and the fluvial gravel units of the Ringold Formation. The uppermost aquifer system is generally under unconfined hydraulic conditions throughout most of the study area. However, in the southwest portion of the 200 East Area, the fine-grained lower mud sequence of the Ringold Formation acts as a confining layer within the uppermost aquifer (Graham et al. 1984).

To the south and west of the 200 East Area, the water table is generally contained within the Ringold Formation. The water table is located near the Hanford/Ringold contact in the central and southern portion of the 200 East Area. However, in the northern portion of the 200 East Area and within the study area, the water table is predominantly contained within the Hanford formation.

The saturated thickness of the uppermost aquifer is dependent on the geologic structure. Approximately 4 km (2.5 mi) south of the 200-BP-1 operable unit, along the axis of the Cold Creek syncline, the uppermost aquifer is over 100 m (330 ft) thick. Along the flanks of bordering anticlinal structures to the north, the aquifer thins to zero. Directly beneath the 200-BP-1 operable unit and in the area to the north (downgradient), the uppermost aquifer system is very thin, ranging from only about 1 to 5.5 m (3 to 18 ft) in thickness. Further to the north, in the vicinity of the erosional window and Gable Mountain/Gable Butte gap, the uppermost aquifer thickens again to approximately 20 to 30 m (65 to 100 ft). In this area, the Elephant Mountain Member is not present and the base of the uppermost aquifer is defined by the underlying Pomona Basalt Member of the Saddle Mountains Basalt Formation. The aquifer in this area is comprised of sediments of the Rattlesnake Ridge interbed, as well as Hanford formation deposits. The thickness of the uppermost aquifer throughout the study area is depicted in Figure 3-38.

3.6.2.2.2 Recharge. Sources of recharge to the unconfined aquifer in the study area are generally similar to those described previously for the regional system, i.e., precipitation and infiltration of runoff to the water table, principally around the periphery of the basin and from small ephemeral streams. As discussed in Section 3.2 (Meteorological

Characteristics) little, if any natural recharge to the groundwater occurs in the broad flat plain of the study area because of the high rates of evapotranspiration. Many areas within the boundary of the 200 East and West Areas are devoid of vegetation and may provide increased recharge relative to vegetated areas.

Artificial recharge to the groundwater in the form waste disposal is the major form of recharge in the separations areas. In the 200 East Area, disposal activities at B pond continue to exert a major influence on groundwater flow in the study area.

An additional source of recharge to the unconfined aquifer may be upward leakage from the lower basalt aquifers. As described below, the vertical gradient between confined aquifers of the underlying basalts and the unconfined aquifer, with some exceptions, is generally upward beneath the 200-BP-1 operable unit and throughout the study area. Vertical gradients are discussed further in Section 3.6.2.2.4 (Vertical Gradients/Aquifer Communication).

3.6.2.2.3 Movement. As discussed in Section 3.6.1 (Regional Hydrogeology), the regional direction of groundwater flow in the uppermost aquifer is from the recharge area on the western side of the Hanford Site to the Columbia River north and east of the operational areas. This general west to east flow pattern is interrupted locally by the artificial recharge which has occurred in the 200 Areas. Groundwater mounds are evident below the B pond and U pond (deactivated in 1984) and occurred, to a lesser extent, beneath Gable Mountain Pond when it was active. The effects on the groundwater flow system from these mounds has been widespread. Water table elevation contours depicting the flow of groundwater in the study area in December 1991 are presented in Figure 3-39.

Newcomer (1990) reports that the major influence on the current configuration of the water table in the study area is the groundwater mound located in the vicinity of the 216-B-3 pond system (B pond). B pond is located east of the 200 East Area and consists of a main pond and three interconnected lobes for wastewater disposal. The pond began receiving liquid effluent in 1945 and currently covers approximately 14 ha (35 ac), with a maximum depth of about 6 m (20 ft) (WHC 1991). Artificial recharge of large volumes of wastewater from B pond has elevated the water table surface in the east and southeast of the 200 East Area, but at present is declining. Groundwater flow is from west to east in the area between the 200 East and West Area, radially outward from B pond, and generally to the north between Gable Mountain and Gable Butte in the area north of 200 East.

The direction of groundwater flow within the 200 East Area vicinity, but outside the B pond mound area, is difficult to determine from water table elevations because of a lack of a pronounced horizontal hydraulic gradient. The horizontal gradient has been reported to be very slight (0.0001 to 0.0002) (Connelly et al. 1992). Contaminant plumes, which will be discussed further in Section 4, indicate that the flow direction in the northern part of the 200 East Area is to the north (Connelly et al. 1992). This pattern of flow is generally evident in the water table elevation data presented in Figure 3-39 for the Gable Gap area.

Data presented in Figure 3-35 indicate that water table elevations beneath the 200 Areas have been declining since 1988. This general lowering has occurred despite continued use of B pond for wastewater disposal. The decline is attributed to the closure of the Gable Mountain pond and U pond, and the general decrease in disposal of separations area process water to B pond (Kasza et al. 1992).

The future configuration of the water table within the study area depends on the future disposal volumes at B pond. If discharges to B pond continue to decline or are eliminated, it is likely that the observed decline in water levels will continue and the regional west to east flow pattern will probably reassert itself and predominate over most of the 200 East Area.

3.6.2.2.4 Vertical Gradients. Limited information relating to the occurrence of vertical hydraulic gradients within the uppermost aquifer was presented in Connelly et al. (1992). Data from nested well groups in the study area were evaluated to determine the presence, magnitude and direction of vertical gradients within the uppermost aquifer. In the vicinity of B pond, water levels in wells 699-43-42J and 699-42-42B, which monitor the upper and lower portions of the unconfined aquifer, indicated a downward gradient of approximately 0.07. At another well nest adjacent to B pond, comprised of wells 699-43-41E, F and G, a downward gradient was also indicated. The three wells monitor different portions of the semi-confined aquifer (Ringold Formation). The downward vertical gradient in the vicinity of B pond is caused by the groundwater mound within the unconfined system which has resulted from disposal of effluents.

Data available from other nested wells located away from the B pond groundwater mound (near the grout treatment facility), indicated that the vertical head differences were slight and difficult to ascertain (Connelly et al. 1992). At wells 299-E25-29P and 29Q, 299-E25-30P and 30Q, 299-E25-32P and 32Q, and 299-E2534 and 299-E25-28, the measured head differences were so slight as to be indistinguishable from measurement errors and therefore inconclusive. In the 699-53-55A, B and C well group, however, which is located in the "erosional window" area north of the 200 East Area, a downward hydraulic gradient was reported. The three wells monitor the upper to middle portions of the uppermost aquifer and are all screened within the Hanford formation. A head difference of approximately 0.09 m (.3 ft) was indicated between the upper and lower screened intervals, a vertical distance of approximately 21 m (70 ft). This results in a downward hydraulic gradient of approximately 0.004.

The results of the evaluation indicated, therefore, that in general, downward vertical gradients in the uppermost aquifer occur in the vicinity of B pond; however, in other portions of the study area outside of the B pond groundwater mound, the nature of the vertical gradients within the uppermost aquifer are uncertain because of limited data. In the area of the 699-53-55 well cluster, however, measurable vertical gradients do occur and are directed downward.

The cause of the downward gradient within the Hanford formation in the vicinity of the 699-53-55 well cluster is unclear. Information presented in Swanson (1992), however, indicates that the observed downward movement of groundwater is confined to the sediments of the Hanford formation and does not extend to the deeper basalt aquifers. The gradient between well 699-53-55A and 699-53-55AP, a fourth well located at the 53-55 well cluster and which is screened in the Pomona basalt, is directed upward. It is considered possible that the cause of the downward gradient within the Hanford formation is related to the geometry of the flow system in the vicinity of the erosional feature, and/or to the presence of a fine-grained interbed. An analysis of the presence and possible cause of the downward gradient at the 699-53-55 well cluster, using groundwater modeling techniques, is presented in Section 3.6.3 (Analysis of Observed Downward Hydraulic Gradients at Well Cluster 699-53-55).

3.6.2.2.5 Discharge. Discharge of groundwater from the uppermost aquifer occurs primarily to the Columbia River. In the northern portion of the study area, which includes the 200-BP-1 operable unit, groundwater flow is to the north through the Gable Mountain and Gable Butte area to the Columbia River. In the southeast portion of the 200 East Area groundwater flow occurs to the southeast (Figure 3-39).

West Lake represents a likely discharge location for the uppermost aquifer. The lake is a topographic depression that intersects the water table. Because of high surface water evaporation rates and low surface overland flow, the lake probably represents a local discharge zone.

Leakage may also be occurring to underlying confined aquifers in areas where vertical gradients are directed downward, such as in the vicinity of the groundwater mound beneath B pond. Aquifer communication between the unconfined and confined aquifers will be discussed further in Section 3.6.2.4.

3.6.2.2.6 Hydraulic Properties. The results of a series of hydraulic tests performed in wells of the uppermost aquifer were compiled and summarized in Connelly et al. (1992). The results include estimates of hydraulic properties obtained from 29 constant discharge/recovery tests which were performed at wells throughout the 200 East aggregate area. Test data were obtained primarily from Newcomer et al. (1992a) and Swanson (1992). Swanson (1992) includes descriptions of well tests, including constant discharge, step-drawdown recovery, and slug tests, which were performed at three unconfined aquifer wells tested as part of the 200-BP-1 Phase I RI. These include wells 699-52-54, 699-52-57, and the 699-53-55 well group.

The majority of the tests summarized in Connelly et al. (1992) were performed as single well tests in partially penetrating wells. Table 3-7 summarizes the transmissivity and hydraulic conductivity values which were obtained for the uppermost aquifer.

Based on the estimates of hydraulic conductivity presented in Table 3-7, a map was constructed of hydraulic conductivity contours for the study area (Connelly et al. 1992). This contour map is depicted in Figure 3-40. Because the map is constructed from a limited number of data points, it should be used as a general indication only of the hydraulic conductivity distribution throughout the study area. Overall, hydraulic conductivities (K) for the uppermost aquifer range from approximately 10^{0} to 10^{-3} cm/s (10^{3} to 10^{0} ft/d).

Based on the map presented in Figure 3-40, there appears to be two distinct zones of relatively high K within the study area. One is located north of the basalt subcrop in the vicinity of wells 699-53-55 and 699-55-50. The second zone occupies the central portion of the 200 East Area and trends generally from the northwest to the southeast. Values of hydraulic conductivity in these two areas are in the range of 10^{-1} to 10^{1} cm/s (10^{2} to 10^{4} ft/d). Lower values of K in the study area occur to the south and west. In Connelly et al. (1992), the areas characterized by high K values were generally associated with the occurrence of Hanford gravel while low K zones were correlated with occurrences of Ringold Units.

Swanson (1992) reported a qualitative specific yield of 0.46 for well 699-53-55C. This value is beyond the upper range expected for unconfined aquifers (Freeze and Cherry 1979). The high specific yield may reflect in part the high estimated transmissivity for the well and the open framework gravels present at the well location.

In addition to the values presented above, which were based primarily on measurements made in constant-discharge pumping tests, slug test data are available from each of the 7 unconfined aquifer wells installed under Task 6 of the work plan. The results of these tests are summarized in Hoffman (1992). Locations of the wells are depicted in Figure 2-5. Values obtained from these tests were generally consistent with the range of values presented above for the unconfined aquifer.

3.6.2.3 Confined Aquifer. The uppermost regionally confined aquifer in the study area is the Rattlesnake Ridge aquifer. The aquifer is separated from the overlying unconfined aquifer of the Hanford formation by the Elephant Mountain Member of the Saddle Mountains Basalt Formation. To the north of the 200-BP-1 operable unit, the Elephant Mountain basalt has been eroded and the Rattlesnake Ridge interbed is in direct contact with the uppermost aquifer system. This occurs in the area of the "erosional window" associated with monitoring wells 699-53-55 and 699-55-55 (Hoffman et al. 1992). This may also occur in the Gable Gap area (along the crest of the Umtanum Ridge-Gable Mountain anticline) where erosion has occurred down through at least the Pomona basalt. Because the Rattlesnake Ridge interbed has been removed in this area as well, the extent of the direct contact between the uppermost aquifer and the Rattlesnake Ridge aquifer is unclear in the Gable Gap area. The approximate locations of the erosional features in the Elephant Mountain basalt and Rattlesnake Ridge interbed are depicted in Figures 3-27 and 3-28, respectively.

3.6.2.3.1 Recharge. Recharge of the Rattlesnake Ridge aquifer is felt to occur primarily along the higher elevations surrounding the basin to the west, north, and northeast (Graham 1983). In addition, as described below, artificial recharge is also probably occurring in areas of water table mounding which have resulted from the disposal of large volumes of waste waters in the 200 Areas. These areas include B pond, and the 200 West Area (Jackson 1992).

3.6.2.3.2 Movement. Groundwater flow in the Saddle Mountains basalt aquifers in the Hanford Site vicinity is felt to occur predominantly from west to east (Gephart et al. 1979). In the vicinity of B pond, however, observed flow in the Rattlesnake Ridge aquifer is to the west-northwest (Graham 1983; Kasza et al. 1991; and Jackson 1992). The groundwater mound which has developed in the unconfined aquifer beneath B pond has apparently led to the creation of a localized disturbance to the general west to east groundwater flow pattern in the Rattlesnake Ridge aquifer.

Kasza et al. (1991) presented a potentiometric map for the Rattlesnake Ridge aquifer in the vicinity of B pond. This map, which is depicted in Figure 3-41, is produced annually to examine the potential interaction between the confined and unconfined aquifers in the vicinity of B pond. Kasza et al. (1991) reported that, based on this information, the groundwater flow direction north and west of the 200 East Area is northward towards the Gap between Gable Mountain and Gable Butte. East and south of the 200 East Area the flow is generally directed to the east.

Information presented in Graham (1983), which included a more site-wide potentiometric surface map for the Rattlesnake Ridge aquifer, suggested that flow in the Rattlesnake Ridge aquifer is directed towards the Gap Area from recharge areas located to the west, north, and east. Graham (1983) further suggests that the Gap area may represent a groundwater discharge zone for the Rattlesnake Ridge aquifer to the unconfined aquifer.

The most recent interpretation of groundwater flow in the Rattlesnake Ridge aquifer is presented in Jackson (1992). The 200-BP-1 Phase 1 RI measured water levels in the confined aquifer in the study Area. Figure 3-42 presents a potentiometric surface from water levels measured April 1991 for the Rattlesnake Ridge aquifer. The map indicates groundwater in the Rattlesnake Ridge confined aquifer is moving from B-pond toward the Gable Mountain Gap Area in the study Area. This interpretation supports the regional view that the Gap area represents a groundwater discharge zone for the Rattlesnake Ridge interbed.

3.6.2.3.3 Discharge. On a regional basis, the confined aquifers of the Saddle Mountains Basalts are believed to generally flow from west to east and discharge to the unconfined aquifer in the topographically low regions of the plateau and ultimately to the Columbia River (DOE 1988). Hydraulic head data presented in Figure 3-37 suggest that the dominant potential discharge area appears to be along a broad region bordering the Columbia River to the east of the 200 East Area. At the study area scale, however, the issue is complicated by the presence of major structural and erosional features near the Gable Mountain-Gable Butte area, and groundwater mounding in the unconfined aquifer.

Graham et al. (1984) suggests that in the vicinity of West Lake, where the Elephant Mountain basalt and Rattlesnake Ridge interbed have been eroded, the Rattlesnake Ridge aquifer discharges to the unconfined aquifer. This view is based upon apparent upward gradients measured between well 699-59-55 and West Lake. The confined waters discharged to the unconfined aquifer move within the unconfined aquifer and eventually discharge to the Columbia River. This view is generally consistent with the potentiometric data presented in Figure 3-37 which depicts the Gap area as a possible hydraulic sink for the Rattlesnake Ridge aquifer.

DOE (1988) reports that, based primarily on available areal and vertical hydraulic head relationships for the Mabton interbed of the Saddle Mountains Basalt, groundwater in the Saddle Mountains Basalt discharges to the unconfined flow system and the Columbia River except across the northern portion of the Hanford Site. In the Umtanum Ridge-Gable Mountain area, a potential discharge zone is noted due to observed low hydraulic heads. This discharge zone is believed to be attributable to enhanced vertical hydraulic communication within the anticlinal structural area. While this interpretation is made for the Mabton rather than the Rattlesnake Ridge, it is essentially consistent with the interpretations presented above (and in Figure 3-37) reflecting the likelihood that the Gap area serves as a hydraulic sink for the Rattlesnake Ridge aquifer.

3.6.2.3.4 Hydraulic Properties. Hydraulic properties for the confined aquifer have been compiled and summarized by Connelly et al. (1992). This information is presented in Table 3-8. The test results are derived from constant discharge pumping tests summarized in Newcomer et al. (1992b) and include the recent hydraulic testing which was performed (at well 699-49-57B) under the 200-BP-1 operable unit work plan (Swanson 1992). The hydraulic properties presented in Table 3-8 were measured in 17 confined aquifer wells, 15 of which were installed within the Rattlesnake Ridge interbed. Estimates of hydraulic conductivity for the Rattlesnake Ridge interbed range from 1.4×10^{-4} to 7×10^{-3} cm/s (0.4 to 20 ft/d) and average approximately 3×10^{-3} cm/s (9 ft/d). These values generally fall within the range reported in Section 3.6.1 (Regional Hydrogeology) for the basalt interbeds. The value at well 699-49-57B, which was measured under the 200-BP-1 operable unit RI, was 5×10^{-3} cm/s (14 ft/d).

In addition to these estimates, slug test data are available from each of the 3 confined aquifer wells installed under Task 6 of the work plan. The results of these tests are summarized in Hoffman (1992). Locations of the wells are depicted in Figure 2-5. Values obtained from these tests were generally consistent with the range of values presented above.

3.6.2.4 Vertical Gradients/Aquifer Communication. Aquifer communication is a process whereby groundwaters between distinct hydrogeologic systems are transferred and mix. Beneath the 200-BP-1 operable unit, hydraulic heads are higher in the confined Rattlesnake Ridge interbed aquifer than in the overlying unconfined aquifer. This upward vertical gradient below the 200-BP-1 operable unit indicates that mass flux (leakage) of groundwater is currently from the Rattlesnake Ridge aquifer to the unconfined aquifer. For the purposes of this report, aquifer communication will refer to the migration of groundwater from the unconfined aquifer to the underlying confined aquifer as evidenced from downward vertical hydraulic gradients. The significance of the process for this study lies primarily in the potential for contamination of the Rattlesnake Ridge aquifer, and possibly of other deeper basalt interbed aquifers, by contaminated groundwater of the uppermost aquifer system. Contamination of the confined aquifer systems is of concern because of the relative lack of hydrogeologic understanding which has been developed with respect to groundwater movement, flow directions and discharge. There is the potential, because of the greater depth of these systems and the more regional nature of their flow, for movement of contaminated groundwater to discharge points outside of the Hanford Site boundaries.

This section will examine the available data which has been generated regarding the possible occurrence of aquifer communication in the study area. It includes data which has been presented in previous studies, as well as an evaluation from a recent study which was performed specifically for the 200-BP-1 operable unit Phase I RI. These studies have included evaluations of a variety of data types and sources, including:

- Characterization of possible geologic and structural mechanisms which could contribute to or permit aquifer communication, such as the presence of erosional features, fault and joint systems
- Evaluations of hydraulic head data in clustered wells to determine the presence or absence of downward vertical gradients between aquifer systems
- Contaminant trends in Rattlesnake Ridge aquifer wells which could indicate possible past and present occurrences of contamination from the overlying aquifer
- Potentiometric data which could provide indications of hydraulic continuity between shallow and deep aquifer systems
- Miscellaneous information such as borehole construction details which could provide explanations of localized aquifer communication.

3.6.2.4.1 Previous Studies. Evidence of communication between the uppermost aquifer system and the Rattlesnake Ridge aquifer has been documented previously by Graham et al. (1984), Jensen (1987), and DOE (1988). DOE (1988) provided a general discussion of vertical gradients and possible aquifer communication in areas across the

Hanford Site. Leakage between the two systems was reported to occur in erosional areas that are expected to represent localities of substantial vertical hydraulic communication (e.g., Gable Mountain/West Lake/B pond area) and through distributed leakage, wherever vertical hydraulic gradients exist. Based on field head measurements, the eastern half of the Hanford Site was described as an area of generally upward hydraulic gradients between the confined and unconfined systems, while the western half (including the Gable Mountain/West Lake/B pond area) was described as an area of predominantly downward groundwater flow. This conclusion, however, was based upon the collection of no head measurements in the Gable Mountain/West Lake/B pond vicinity. Interpretation that the flow is uniformly downward in that portion of the Hanford Site, therefore, must be viewed with caution.

DOE (1988) also evaluated reported I-129 levels which occur above background in the Mabton and Rattlesnake Ridge aquifers. While the means of the I-129 introduction into the confined aquifers was not fully defined, the Umtanum Ridge-Gable Mountain anticline was identified as a likely source of vertical groundwater leakage. It was suggested that high hydraulic heads beneath B pond and Gable Mountain pond in the unconfined aquifer could have created sufficient driving force to transport groundwater potentially contaminated with low-level radionuclides from the unconfined aquifer to the basalt aquifers. In addition, three large water cooling basins, located about 3.2 km (2 mi) northwest of the 200 East Area, were identified as an additional potential contaminant source. Records indicate that up to 1×10^9 L (264×10^6 gal) of water were disposed of in that area (DOE 1988).

Graham et al. (1984) focused specifically on assessing locations and extent of aquifer communication in the area surrounding B pond and Gable Mountain pond, and encompassing the 200 East Area where subsurface liquid-waste disposal facilities are located. Two areas of complete erosion of the Elephant Mountain basalt were identified in the study area. Two other areas were inferred from geological and hydrological (barometric) evidence. Downward gradients from the unconfined aquifer to the Rattlesnake Ridge aquifer were identified in the immediate vicinities of Gable Mountain pond and B pond. These downward gradients did not extend to the known or suspected areas of erosion of the Elephant Mountain basalt at the time of the study.

From the groundwater chemical data, an area where aquifer communication had occurred was identified south and east of Gable Mountain pond extending to the 200 East Area. It was suggested that this communication probably occurred in the late 1960s and early 1970s when the water table was at a higher elevation. As a result of the aquifer communication, low levels of tritium and I-129 contamination were identified in the Rattlesnake Ridge aquifer. In addition, migration by density flow down the annular space of borehole 699-E33-12 (when it was apparently uncased and open to both aquifers) was identified as another significant mechanism of aquifer communication. The study reported that contamination in the Rattlesnake Ridge aquifer will eventually discharge back to the unconfined aquifer in the vicinity of West Lake due to the presence of upward gradients.

Jensen (1987) consisted of a review of hydraulic head and chemical data to determine the extent of communication between the unconfined and Rattlesnake Ridge aquifers in the B pond-Gable Mountain area. The study concluded that the presence of chemical constituents in the Rattlesnake Ridge aquifer confirmed that aquifer communication had occurred in the area. The mechanism was proposed to be downward leakage through erosional windows and fractures in the Elephant Mountain basalt in the area of downward

vertical gradients associated with the B pond. Discharge of contaminants from the Rattlesnake Ridge aquifer was interpreted to occur back to the unconfined aquifer in the vicinity of West Lake where erosion of the overlying Elephant Mountain Basalt had occurred.

3.6.2.4.2 Current Evaluation of Aquifer Communication. The potential for aquifer communication has decreased in the 200 East Area and vicinity because of a decrease in liquid waste disposal operations which have occurred due to the termination of various plant operations (Connelly et al. 1992). Water levels in the uppermost aquifer system have generally declined, thereby reducing, but not eliminating, the extent and magnitude of downward hydraulic gradients. Also, the disposal of the high density, high-salt wastes was terminated. Although the potential for aquifer communication has been lessened, it is likely that some areas within the 200 East Area vicinity are still conducive to the occurrence of the process. This subsection will focus on assessing the current occurrence of aquifer communication in the 200 East Area.

Site characterization objectives of the 200-BP-1 operable unit Phase I RI required an assessment of aquifer communication in the northern portion of the 200 East Area and to the north. As a part of this assessment, a series of field activities were undertaken to determine geologic and hydrologic conditions of the uppermost aquifer system and the Rattlesnake Ridge confined aquifer. These included the installation of additional wells, the refurbishment of several existing wells, and the establishment of a series of five well clusters to allow observation and comparison of hydraulic heads and groundwater quality in the two aquifer systems. These well clusters are listed in Table 3-9 and are depicted in Figure 3-43.

Connelly et al. (1992) utilized data collected from these well clusters, as well as additional hydrogeologic data which has been collected since the studies of Graham et al. (1984) and Jensen (1987) to assess the current extent and magnitude of aquifer communication in the 200 East Area. The following discussion summarizes the assessment of Connelly et al. (1992).

The evaluation of aquifer communication performed in Connelly et al. (1992) was based primarily on the following data analyses:

- **Groundwater quality as an indicator of aquifer communication.** The levels of certain indicator constituents (Tc-99, Co-60, I-129, and tritium) in Rattlesnake Ridge groundwater were evaluated with time (trend analyses) to qualitatively assess the general localities where aquifer communication has occurred in the past and may be occurring presently.

- **Barometric efficiencies.** The barometric efficiencies of the Rattlesnake Ridge aquifer cluster wells were determined. This information was then used as a qualitative indicator of the presence of erosional windows in the Elephant Mountain basalt, as was done in Graham et al. (1984). Near erosional windows, the Rattlesnake Ridge aquifer is reportedly less influenced by atmospheric pressure effects since the upper confining layer is absent; thus, the barometric efficiency of wells installed near erosional windows will be less than for wells installed in a truly confined area.

- **Hydraulic head comparisons.** The magnitude and direction of the vertical hydraulic head component was determined at the well clusters. Rattlesnake Ridge aquifer hydraulic head values were corrected for atmospheric pressure effects using the barometric efficiency data.

  The results of the evaluation indicated that aquifer communication between the uppermost aquifer and the Rattlesnake Ridge confined aquifer does appear to be an active process in the 200 East Area but that it is currently limited to the vicinity of the B pond. A substantial downward vertical gradient exists beneath the pond due to the development of a large groundwater mound in the uppermost aquifer system. The approximate areal extent of downward vertical gradients is depicted in Figure 3-43. Increases in tritium concentrations at well 699-42-40C, which is located beneath the B pond groundwater mound, indicate that groundwater from the uppermost aquifer system is currently being introduced to the Rattlesnake Ridge aquifer.

  Head trends in the uppermost aquifer system were typically mirrored closely by head trends in the Rattlesnake Ridge aquifer, based on an analysis of head data collected at the well clusters. This trend similarity indicates that the two aquifer systems are hydraulically interconnected and continuous (Jensen 1987). If vertical gradients were directed downward, contamination of the Rattlesnake Ridge aquifer could potentially occur because of the interconnectivity. Vertical gradients at the well clusters, however, as indicated in Table 3-9, currently are directed upward in every case. This suggests that flow of groundwater in the northern portion of the 200 East Area and to the north is currently from the Rattlesnake Ridge aquifer to the unconfined aquifer. The maximum head differential (upward gradient of 0.005) was reported for the 299-E33-40/299-E33-07 well cluster. Other well clusters exhibited more modest vertical gradients.

Groundwater quality data from Rattlesnake Ridge aquifer wells demonstrate that aquifer communication has occurred in the past at a number of locations in the vicinity of the 200 East Area. The suspected locations include the following:

- **Monitoring well 699-42-40C.** This well is located in close proximity to the B pond. Contamination detected in the well is thought to be the result of B pond operation and the resulting creation of a groundwater mound in the unconfined aquifer. The activity trend plot for the well indicates currently increasing tritium levels indicating that contaminated groundwater from the uppermost aquifer continues to be introduced to the Rattlesnake Ridge aquifer under a downward hydraulic gradient. The flow path mechanism is not clearly understood but is thought to consist of leakage through the Elephant Mountain Basalt in fractures and/or joint systems, or through an undetected erosional window.

- **Monitoring Well 699-47-50.** Groundwater quality data from monitoring well 699-47-50 provides historical evidence that past aquifer communication has occurred in the vicinity. Currently, tritium levels at the well are decreasing, suggesting that the downward flux of contaminants from the unconfined aquifer is reduced or no longer operative. Both Graham et al. (1984) and Jensen (1987) report indirect evidence for the presence of an erosional window in the vicinity of the well which could serve as a pathway for contaminant

movement into the Rattlesnake Ridge. No direct geologic evidence of the feature has been collected, however. It is equally possible that the presence of significant fractures or joint systems in the basalt could also permit aquifer communication to occur.

- **Monitoring well 699-54-57.** Groundwater quality data from the well indicate that aquifer communication has occurred in the vicinity over most of the well's sampling history. The mechanism of communication is thought to be associated with the presence of the erosional feature located immediately to the east. The uppermost aquifer system in the area of the erosional window contains substantial concentrations of man-made radionuclides associated with waste disposal. Recent data indicates that levels in the Rattlesnake Ridge are currently diminishing. This may be due to decreases in head in the uppermost aquifer, thus reversing the direction of vertical flow from downward to the current upward flow pattern, or to a reduction in the flux of contaminants entering the confined system.

- **Monitoring Well 299-E33-12.** The well was drilled in the mid-1950s and left uncompleted until 1982, thereby acting as a direct vertical conduit between the uppermost and confined aquifer systems. This hydraulic connection in turn may have allowed a dense high-salt waste plume (probably originating from the BY cribs) to migrate downward via the well's annular space (Graham et al. 1984). Analyses performed since 1982 indicate that activity levels have decreased substantially since the well was completed. Recent increases at the well location are felt to be related to new sampling techniques which more effectively purge the well prior to sample collection and not to the occurrence of aquifer communication (Connelly et al. 1992).

- **Monitoring well 299-E26-8.** The well exhibited elevated tritium levels which then fell off rapidly during the 1984-1985 period. The cause of the elevated levels and subsequent rapid decline is not clearly understood. No geologic features are known to exist nearby which could act as contaminant pathways. The well is located downgradient from the portion of the Rattlesnake Ridge aquifer which is potentially being impacted beneath the B pond groundwater mound. It may be that contamination detected at the well migrated from the vicinity of B pond. Another possibility is that the tritium levels were introduced into the well as a result of construction techniques. Tritium levels have decreased below detection limits since 1985.

3.6.2.4.3 Summary of Aquifer Communication. Based on the results of the above referenced studies which have been conducted to asses the extent and nature of aquifer communication in the vicinity of the 200 East Area, the following summarizing conclusions can be made:

- Aquifer communication, defined in this case as the movement of contaminated groundwater from the uppermost aquifer to the underlying Rattlesnake Ridge confined aquifer, has occurred throughout the vicinity of the 200 East Area for at least several decades, at several locations, and due to a number of physical mechanisms.

- The primary mechanism leading to the occurrence of aquifer communication has been the occurrence of downward hydraulic gradients caused by groundwater mounding in the unconfined aquifer beneath B pond and possibly Gable Mountain Pond. The downward gradients have occurred in association with erosional features (and possibly fracture/joint systems) to result in downward migration of contaminants from the uppermost to the confined aquifer system.

- There are at least one and possibly two erosional features where direct aquifer communication has occurred in the past. These include one confirmed erosional window to the north of the 200 East Area (in the vicinity of wells 699-53-55 and 699-55-55) and one suspected erosional feature at the northeast corner of the 200 East Area. While these features are thought to have allowed communication to occur in the past when gradients were suspected to be downward, they are not currently felt to be operative areas of aquifer communication. The area of downward gradients is not as extensive as it is thought to have been in the past, and currently hydraulic gradients between the unconfined and confined aquifer systems are generally upward in the study area except for in the vicinity of the B pond. Contaminant levels at wells near the erosional features are declining.

- The decrease in the areal extent of downward hydraulic gradients is associated with a general decrease in water levels in the uppermost aquifer which has taken place in the last several years. This decline in the water table elevation has resulted from a general decrease in the discharge of liquid wastes in the 200 East Area. Downward gradients are still present beneath the B pond, and aquifer communication is still occurring in the immediate vicinity of the disposal pond.

- Aquifer communication has also occurred at well 299-E33-12 when it remained uncased from the mid-1950s to 1982. During that time period the well acted as a direct hydraulic conduit for contaminated groundwater between the unconfined and confined aquifers.

- While the flow directions and discharge locations of the Rattlesnake Ridge aquifer are not clearly understood, the available evidence suggests that groundwater flow in the B pond/Gable Mountain/West Lake area is currently to the northwest under the influence of the B pond groundwater mound. Discharge of this groundwater is expected to occur to the unconfined aquifer in the vicinity of the Umtanum Ridge-Gable Mountain structure which acts as a hydraulic low and discharge point for the Rattlesnake Ridge and deeper interbed aquifers. Groundwater of the unconfined aquifer then flows generally to the east where it discharges to the Columbia River.

3.6.3 Analysis of Observed Downward Hydraulic Gradients at Well Cluster 699-53-55

Information presented above with regards to vertical gradients generally indicate the following:

- Within the unconfined aquifer, gradients are directed downward in the vicinity of B pond.

- Elsewhere in the study area, vertical gradients within the unconfined aquifer are slight and difficult to ascertain. At wells 299-E25-29P and 29Q, 299-E25-30P and 30Q, 299-E25-32P and 32Q, and 299-E25-34 and 299-E25-28, the measured head differences were so slight as to be indistinguishable from measurement errors.

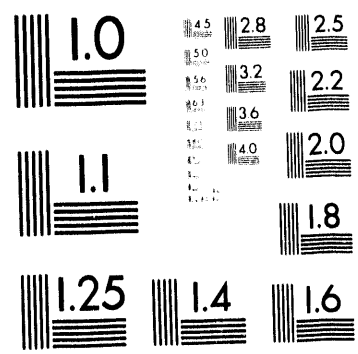
- Between the confined and unconfined aquifers, gradients are generally upward throughout the study area, except for in the vicinity of B pond (Table 3-9 and Figure 3-43).

- In the 699-53-55A, B and C well group, however, which is located in the "erosional window" area north of the 200 East Area, a downward hydraulic gradient was reported. The three wells monitor the upper to middle portions of the uppermost aquifer and are all screened within the Hanford formation. A head difference of approximately 0.09 m (0.3 ft) was indicated between the upper and lower screened intervals, a vertical distance of approximately 21 m (70 ft). This results in a downward hydraulic gradient of approximately 0.004. The majority of the head difference (0.08 m [0.25 ft]) occurs between the middle and lower well. The gradient between these wells is approximately 0.003.

The downward vertical gradients at the 699-53-55 well cluster are anomalous with respect to head data collected elsewhere in the study area. It is of concern because of the potential for downward movement of groundwater from the unconfined to the confined aquifer.

Discussed below is a finite-element model of groundwater flow in the erosional window. This model is used as a tool to better understand the conditions which might result in the apparent anomaly of a downward hydraulic gradient at the 699-53-55 well cluster. Simulations were run using the Golder Groundwater Package, a two dimensional finite element flow and transport code. The modeled conditions include: the geometry of the base of the unconfined aquifer, hydraulic conductivity anisotropy, hydraulic conductivity heterogeneity, and combinations of the above. The finite-element mesh and material zones for the model are presented in Figure 3-44. The southeastern (right) boundary of the model corresponds to well 699-52-54, and the northwestern boundary to well 699-54-57. The shape of the grid and material contacts are based on the cross-section presented in Figure 3-22. Those nodes labeled A, B, and C in Figure 3-44 correspond to the mid-points of the screened-intervals for wells 599-53-55A, 699-53-55B, and 699-53-55C, respectively.

A total of five model simulations were conducted by varying the hydraulic conductivities of the six material zones shown in Figure 3-44. Material zone horizontal and vertical hydraulic conductivities for each simulation are presented in figure 3-45. Vertical hydraulic conductivity is assumed to be five times less than horizontal hydraulic conductivity for sands and gravels, and ten to one-hundred times less for fine-grained sediments. In all five simulations material zone one (1) corresponds to the Rattlesnake Ridge Interbed, zone three (3) corresponds to the Elephant Mountain Basalt, and zones four through six correspond to the Hanford formation. Material zone two (2) corresponds


1.0
1.1
1.25
1.4
1.6
1.8
2.0
2.2
2.5
2.8
3.2
3.6
4.0
4.5
5.0
5.6
6.3

2 of 6

to the Elephant Mountain Basalt in simulations A through D, and the Hanford formation in simulation E. Note that the geometry of the erosional window is of limited extent parallel to groundwater flow in simulations A through D, but becomes an essentially continuous erosional channel along groundwater flow to the northwest toward the Gable Gap area in simulation E.

Boundary conditions for the model consist of no-flow along the bottom of the Rattlesnake Ridge Interbed, and fixed hydraulic head along the southeast and northwest boundary. Values of hydraulic head were derived from measurements taken in wells 699-52-54 and 699-54-57, and interpolated from groundwater contour maps. For simulations A through D the hydraulic head varies southeast to northwest from 122.9 to 122.7 m (403.1 to 402.6 ft) in the Hanford formation as a model generated phreatic surface, and 123.0 to 122.9 m (403.5 to 403.0 ft) in the Rattlesnake Ridge Interbed as a confined aquifer. In simulation E the Rattlesnake Ridge Interbed is not hydraulically confined along the northwestern boundary because material zone two represents the Hanford formation. As such, the boundary conditions for simulation E differ from the other simulations in that the entire northwestern boundary is given a hydraulic head of 122.7 m (402.6 ft).

Results for all five simulations are presented in Figure 3-45 as the difference in hydraulic head between well 699-53-55B and well 699-53-55C, respectively. Simulations A, B, and C result in no measurable difference in hydraulic head in the two wells. This suggests that the difference in hydraulic head between well 699-53-55B and well 699-53-55C predicted by the model is insensitive to:

- A small decrease in the upward movement of water between the Rattlesnake Ridge Interbed and the Hanford formation due to an increase in the Rattlesnake Ridge Interbed anisotropy (simulation B)

- An increase in the downward movement of water in the upper Hanford formation due to an increase in hydraulic conductivity of the lower Hanford formation (simulation C).

Simulations D and E did result in measurable differences in hydraulic head between wells 699-53-55B and 699-53-55C. The difference in hydraulic heads predicted by the model for simulation D is approximately an order-of-magnitude less than that measured in the field, and approximately one-third for simulation E. These results suggest:

- The presence of a semi-confining layer between the screened intervals of well 699-53-55B and well 699-53-55C (simulation D and E) (supported by the geologic log of well 699-53-55A)

- The geometry of the erosional window more likely resembles a continuous channel toward the Gable Gap area (simulation E). This indicates that the erosional feature is not limited in extent along the groundwater flow path but is continuous with the erosional geology which has been observed in the Gable Gap area.

Most importantly simulation E demonstrates that a combination of aquifer geometry, anisotropy, and heterogeneity can explain the anomaly of a downward vertical gradient within the erosional window. More hydrogeologic data is necessary to better define this erosional feature and its influence on the groundwater hydraulics.

3.7 ECOLOGICAL CHARACTERISTICS

This subsection provides a description of potential receptor populations that exist in and around the 200-BP-1 operable unit. Both human and non-human populations are respectively addressed in Sections 3.7.1 (Human Ecology) and 3.7.2 (Wildlife Ecology).

3.7.1 Human Ecology

The description of the human ecology associated with the operable unit focuses on issues pertaining to land use (Section 3.7.1.1), water use (Section 3.7.1.2), and cultural resources (Section 3.7.1.3). Demography is addressed within the land use discussion below.

3.7.1.1 Land Use. Land use, including a discussion of demographics, is presented below. Both regional and local aspects of this topic are addressed.

3.7.1.1.1 Regional Land Use. The Tri-Cities region consists of the incorporated cities of Richland, West Richland, Kennewick, and Pasco and the surrounding communities within Franklin and Benton Counties. Land use in the Tri-Cities region is primarily agricultural, residential, industrial, and recreational.

Most of the agricultural lands are located north and east of the Columbia River and south of the Yakima River. Such lands are used primarily for dry-land and irrigated crop production and livestock grazing. Principal agricultural products include hay, wheat, vegetables (primarily potatoes and corn), apples, grapes and other fruits, and hops (DOE 1987; Jaquish and Mitchell 1988). In 1985, wheat represented the largest single crop in terms of area planted in Benton and Franklin Counties, with 116,000 ha (290,000 ac) (Woodruff et al. 1991). Prior to the establishment of the Hanford Site in the mid-1940s, the majority of what is now the site was used for agricultural purposes, primarily livestock grazing, with some areas used for orchards and irrigated crops.

Residential land use is concentrated around the incorporated areas. Industrial lands are concentrated east of Kennewick along the Columbia River (Benton County Board of Commissioners 1985). Most industrial activities are associated with either agriculture or energy production (DOE 1987).

The Hanford Reach and adjacent wildlife/recreation areas provide a wide variety of recreational activities year round. The most popular activities include sport fishing, flatwater motor boating, and waterfowl hunting. Other popular activities include water skiing, upland bird and deer hunting, and nature observation (NPS 1992).

That portion of the Hanford Site located north of the Columbia River consists of two wildlife reserves - the Wahluke Slope Wildlife Area, a DOW wildlife management area, and the Saddle Mountain National Wildlife Refuge, managed by the USFWS. The northeast slope of the Rattlesnake Hills, along the southwestern boundary of the Hanford Site, is designated as the Arid Lands Ecology Reserve. This reserve is used for ecological research by DOE-RL. It is also designated a National Environmental Research Park (Jaquish and Mitchell 1988). Figure 3-46 shows the locations of these ecological reserves, which cover approximately 45% of the Hanford Site.

The Benton County portion of the Hanford Site is currently zoned as unclassified (Benton County Planning Department undated). Land use is restricted for activities which are associated with the nuclear industry; non-nuclear related activities may be allowed upon approval of DOE (Benton County Code, Title 11, Ordinance No. 62).

The estimated 1990 population totals of Benton and Franklin Counties were 110,000 and 34,600, respectively (Woodruff et al. 1991). For the Tri-Cities, the 1990 estimates were Richland, 30,250; Kennewick, 37,910; and Pasco, 17,820. The populations of Benton City, Prosser, and West Richland totaled 9,615. Population projections for the Tri-Cities had forecast a total growth of 0.09% from 1988 to 1993 (DOE-RL 1990b) but this prediction was made before the 1990 Census, which shows more growth than originally estimated.

3.7.1.1.2 Local Land Use. For reasons of national security, as well as to insure public health and safety, access to the entire Hanford Site has been administratively controlled since 1943. Although DOE is expected to retain control of the Hanford Site in the foreseeable future (DOE-RL 1993), a Hanford Future Site Uses Working Group (1992) is laying out land use alternatives for DOE to consider when it develops an environmental impact statement for the Hanford Site.

Currently, most of the land on the Hanford Site is undeveloped, and the industrial facilities are widely spaced and located in scattered clusters (DOE 1987). Since 1944, most of the land has not been grazed by livestock, all land cultivation and irrigation have ceased, and there has been no resident human population. Recreational land use of the Hanford Site is limited to the bank of the Columbia River up to the high water mark.

Access to most of the 200 Areas, including the 200-BP-1 operable unit, is restricted in addition to the general Hanford Site administrative controls (DOE 1987). Current land use activities associated with the 200-BP-1 operable unit and vicinity are all industrial in nature, relating primarily to fuel reprocessing and waste processing management and disposal activities. Some of the specific activities include waste storage, waste burial, and waste infiltration systems. The 200-BP-4 and 200-BP-3 operable units contain cribs (Figure 3-2). The 200-BP-7 contains the 241-BY, 241-BX, and 241-B Tank Farms. The 200-BP-10 contains burial grounds for solid waste disposal. More specifics on the 200 East Area activities in the vicinity of the 200-BP-1 operable unit, can be found in Section 3.1.3 of this report.

Based on 1980 census data, no one resides within a 1.7-km (1-mi) radius of the 200-BP-1 operable unit. Approximately 110 people live within 17 km (10 mi) of the 200 Areas (DOE 1987); however, the nearest resident is at least 12 km (7 mi) away from the 200-BP-1 operable unit as this is the distance to the nearest Hanford Site boundary. The City of Richland corporate boundary is approximately 30 km (18 mi) to the south. The working population for all shifts in the 200 Areas is approximately 2,400.

3.7.1.2 Water Use

3.7.1.2.1 Surface Water. The Columbia River, which is located approximately 12 km (7 mi) from the 200-BP-1 operable unit at its closest approach, is the most significant surface-water body in the Tri-Cities region and is used as a source of drinking water at onsite facilities and at communities located downstream of the Hanford Site. In addition, the river is used for a variety of recreational activities, including fishing, hunting, boating, water skiing, and swimming (Woodruff et al. 1991).

River water intakes that are downstream from the 200-BP-1 operable unit include the Ringold Fish Hatchery intake, the Ringold Flats irrigation intakes, the Taylor Flats irrigation intakes, the 300 Area process and drinking water intake, the Battelle Farm Operations irrigation intake, the Tri-Cities University Center irrigation intake, and the City of Richland drinking water intake (EPA 1987).

The Hanford Reach of the Columbia River is a popular recreational sport fishing area. Anadromous salmonoids represent the majority of the sport fish harvested. Other significant sport catches include white sturgeon *(Aciponser transmontanus)*, smallmouth bass *(Micropterus dolomieui)*, and walleye *(Stizostedian vitreum)* (DOE-RL 1990b).

Swimming and water skiing are popular recreational activities as well. In the Tri-Cities region both of these activities are centered downstream in the McNary Reservoir. However, a public swimming area has been established at Leslie R. Groves Park, which is approximately 0.8 km (0.5 mi) downstream from the City of Richland water intake (DOE-RL 1990b).

3.7.1.2.2 Groundwater. Groundwater within all aquifers in the immediate vicinity and hydraulically downgradient of the 200-BP-1 operable unit is not used for either drinking or irrigation. The nearest drinking water supply wells are those that serve the 400 Area. They are located about 13 km (8 mi) to the southeast of the operable unit (PNL 1988b; EPA 1988b). However, these wells are not presently downgradient from the 200-BP-1 operable unit.

3.7.1.3 Cultural Resources. The Hanford Site contains an extensive record of past human and animal life, the latter beginning as much as 7 Ma. The Ringold Formation, which underlies the White Bluffs east of the Columbia River, contains one of the most extensive deposits of Pliocene vertebrate fossils in the State of Washington. Remains of extinct camel, horse, rhinoceros, sloth, deer, sabertooth cat, and mastodon are found with bones of still-common turtles and fish in beds of conglomerate and river-deposited clay. Mammoth bones are common in the late Pleistocene silts of the Hanford formation, which cover most western portions of the Hanford Site, and small mammal remains are abundant in Pleistocene and Holocene loess deposits (PNL 1989).

Human beings have lived in the region around Hanford for at least 11,000 years, and by 10,000 years ago had established river-oriented adaptation that was to evolve into the Indian cultures known today. Archaeological sites, some dating back at least 7,000 years and others that are potentially older, are numerous. There are over 250 recorded sites, including the remains of villages with up to 60 houses, fishing camps, game traps, cemeteries, and sites of religious observances. Due to the site access control, most sites have been spared the degree of looting and vandalism that is common elsewhere in the region, and many are in unusually good condition. Fifty-one sites are included in nine properties listed on the National Register of Historic Places, but most of the remaining sites have never been formally evaluated (PNL 1989).

Cultural resources have not been observed in the 200-BP-1 operable unit.

3.7.2 Wildlife Ecology

Wildlife as used in this report, refers to non-domesticated terrestrial and aquatic populations of plants and animals. The description of wildlife ecology associated with the 200 BP-1 operable unit focuses on terrestrial organisms and sensitive habitats.

3.7.2.1 Terrestrial Ecology. The terrestrial ecology of the 200 BP-1 operable unit is described below. Specific studies conducted during the RI were limited and provided little information on the ecological setting of the 200 BP-1 operable unit. Neither flora nor fauna communities were assessed in detail during the RI on the basis of site-specific investigations.

Other studies of the 200 Areas and the 200 Area plateau conducted over the past 20 years were used to define expected plant communities and wildlife associations. These studies were relied upon to provide descriptions of possible wildlife found in the 200-BP-1 operable unit.

Flora

The Pasco Basin is located in the central, semi-arid portion of the Columbia Basin physiographic province, a shrub-steppe vegetational zone. The zone is characterized by the presence of sagebrush and bunchgrass community growing upon desert soils. Daubermire (1970) described the natural climax community of the Hanford Site as a big sagebrush (*Artemisa tridenta*) Sandberg's bluegrass (*Poa Sandbergii*) community within the overall big sagebrush/bluebunch wheatgrass (*Agropyron spicatum*) zonal series.

The 200-BP-1 operable unit is located on the 200 Area plateau. Habitat within the undisturbed areas of the 200 Area plateau consists primarily of the big sagebrush Sandberg's bluegrass community that dominates most of the Hanford Site (Rogers and Rickard 1977). On burned areas of the 200 Area plateau, a cheatgrass Sandberg's bluegrass community predominates, while a cheatgrass/Russian thistle community establishes on recently disturbed soils (Brandt et al. 1991).

The big sagebrush/Sandberg's bluegrass community within the 200 East perimeter fence is dominated by cheatgrass (22–30% cover), with a very low (2-5%) cover by Sandberg's bluegrass (Rogers and Rickard 1977, Brandt et al. 1991). Big sagebrush density is approximately 20 plants/100 m^2 (Brandt et al. 1991). The habitat north of the 200-BP-1 operable unit is more typical of the community type, with 18% cover by Sandberg's bluegrass, 23% cover by cheatgrass, and an average of 18 big sagebrush plants/100 m^2 (Brandt et al. 1991). Plant cover on disturbed sites depends upon the recency of disturbance: unrevegetated waste disposal sites in the 200 East area that had been disturbed 20 years prior to evaluation supported a cover of 66% (rrass and 11% Russian thistle, while sites disturbed less than 10 years supported (han 1% cheatgrass cover and over 40% Russian thistle cover (Rogers and Rickard 1977). Plant species occurring within the 200 Areas were compiled by Price and Rickard (1973) and Cline et al. (1975) and are listed in Table 3-10.

Species protected by Washington State or federal regulations that could occur in the 200-BP-1 area or vicinity include Columbia milkvetch (*Astragalus columbianus*), grey cryptantha (*Cryptantha leucophaea*), and Piper's daisy (*Erigeron piperianus*). These species are primarily associated with sandy soils of the 200 Area plateau and vicinity. Piper's daisy has

been found near B pond outside the 200 East Area (Sackschewsky et al. 1992). The plant surveys of the 200 Areas did not specifically identify any of these species, but a number of milkvetch plants were only identified to genus.

A biological survey of the 200-BP-1 site conducted in August, 1989 (Appendix G) indicated that large portions of the 200-BP-1 operable unit had been disturbed at one time, because the majority of plants occurring in the area were invader species. The major vegetation in the nonradiation area was reported to be rabbitbrush, Sandberg's bluegrass, tumbleweed (Russian thistle) and cheatgrass (Bromus tectorum). Sandberg's bluegrass was the only non-invader. Also observed are aster, mule's ear, sagebrush, goat's beard (yellow salsify), yarrow, and some Indian rice grass. No plants were observed showing signs of stress (no explanation was given as to the criteria used to evaluate stress). No threatened or endangered plant species were observed although Piper's daisy would not have been identifiable at the time the survey was conducted. The results of this survey, as well as a map, are provided in Appendix G.

Plant species observed in the radiation zone were revegetated wheatgrasses (Jiberian = *Agropyron sibericum* and thickspike = *A-dasytachyum*) used to stabilize soil of two cribs. Wheatgrasses on the cribs were reported to be undergoing wind erosion. The major plants around the cribs were cheatgrass, Sandberg's bluegrass, Russian thistle, and rabbitbrush. No threatened or endangered species were observed. However, since the survey was conducted in August it was suggested that a spring survey would be prudent.

Fauna

The only bird observed during the biological survey of the 200-BP-1 area was the western meadowlark. However, bird species observed on the 200 Area plateau are listed in Brandt and Rickard (1991), Fitzner et al. (1992), and Fitzner et al. (1981). A general review of birds observed on the Hanford Site along with their general habitat associations was prepared by Fitzner and Gray (1991). Species associated with the big sagebrush/Sandberg's bluegrass habitat of the 200 Area plateau are listed in Table 3-11, along with designations of federal or State endangered species status.

The ferruginous hawk, Swainson's hawk, loggerhead shrike, and long-billed curlew are candidate species for inclusion on the Federal threatened and endangered species list. Swainson's hawk and the long-billed curlew have been downgraded to Category 3, based on evidence that they are no longer considered seriously threatened throughout their range. Candidates for state listing are the burrowing owl, sage sparrow, and sage thrasher.

Approximately 50 pairs of ferruginous hawks nested in the State of Washington as of 1991 (Fitzner et al. 1992). Ten active ferruginous hawk nests were found on the Hanford Site as of 1991, with eight located in high-tension electric transmission towers and two in trees (Fitzner et al. 1992). These hawks feed primarily on small to medium-sized mammals such as rabbits and ground squirrels (Howard and Wolfe 1976, Fitzner et al. 1981). Two nests occur within 11 km of the 200-BP-1 operable unit (Fitzner et al. 1992).

Loggerhead shrikes are year-round residents on the Hanford Site, although they occur at relatively low densities (Fitzner et al. 1981). They nest from March through August in undisturbed portions of the big sagebrush/Sandberg's bluegrass community, where they average 3.5 pairs/km^2 in the 200 Area plateau (Poole 1992). These medium-sized passerines feed on insects, small mammals, and birds (Fitzner and Rickard 1975). Nestlings in the 200

Area plateau are prey to gophers, snakes, black-billed magpies, common ravens, and coyotes (Poole 1992).

Approximately 15-20 pairs of Swainson's hawks nest on the Hanford Site (Fitzner et al. 1981). The birds nest in trees on the Hanford Site from April to September. Swainson's hawks feed primarily on snakes, medium-sized mammals, and insects, with yellow-bellied racers being the most important prey (Fitzner 1980). The nearest nesting location to the 200-BP-1 operable unit is located just outside the southeastern perimeter fence of the 200 East Area (3.5 km distance). This nest has been in continuous use since at least 1975 (Fitzner et al. 1981).

In 1977, approximately 60 pairs of long-billed curlews were estimated to have been nesting in the 600 Area of the Hanford Site (Allen 1980). Curlews nest from April through June in relatively flat areas dominated by cheatgrass. They feed primarily on beetles and subterranean insect larvae (Allen 1980). Allen (1980) identified a high-use curlew nesting area just east of the 200 East Area within 3 km (1.9 mi) of the 200-BP-1 operable unit. No surveys of the area during the curlew nesting season have been conducted since that time.

Although no burrowing owls were observed during the August (1989) survey of the 200-BP-1 operable unit, badger holes suitable for nests were observed in the area. Burrowing owls are widely distributed on the Hanford Site. The nesting population during the mid-1970s was estimated at 20-26 pairs (Fitzner et al. 1981). Most nest sites are found in abandoned badger and coyote burrows. These small owls are primarily insect and small mammal predators. Insects represent the majority of prey captures, but Great Basin pocket mice (*Perognathus parvus*) form the major part of their diet in terms of biomass (Fitzner et al. 1981). Pocket mice were observed in all parts of the 200-BP-1 operable unit during the August survey.

Sage sparrows are a common summer resident of the 200 Area plateau (Fitzner and Rickard 1975). These small passerines are restricted in their distribution almost entirely to sagebrush stands (Schuler et al. 1988). Sage sparrow abundance on the 200 Area plateau has been shown to be related to sagebrush density (Schuler et al. 1988), although abundance may vary widely between years due to natural environmental variation (Rotenberry 1980). Sage sparrows are the second most abundant bird in the undisturbed areas of the 200 Area plateau, reaching densities of 7.5 birds/km^2 (Schuler et al. 1988). They forage primarily on phytophagous (plant-eating) beetles and other arthropods, with seeds composing less than 5% of their diet (Rotenberry 1980).

The sage thrasher is confined to areas of big sagebrush cover, where it consumes primarily insects and spiders on the ground rather than in the canopy (Terres 1980). Sage thrashers are resident on Hanford Site from spring into the fall (Fitzner and Gray 1991), although at very low densities (Schuler et al. 1988).

The most abundant bird found in the 200 Area plateau is the western meadowlark (Brandt and Rickard 1992). Western meadowlarks are present on the Hanford Site throughout the year (Fitzner and Rickard 1975). These passerines nest on the ground from April through July (Brandt and Rickard 1992). Their diet is composed almost entirely of phytophagous insects (Rotenberry 1980). Meadowlark abundance on the 200 Area plateau was estimated to be approximately 11 birds/km^2 (Schuler et al. 1988).

Red-tailed hawks are the most common hawks nesting on the Hanford Site. At least 20 nesting pairs occupy the site, including several pairs that nest within 3 km of the 200-BP-1 operable unit (Fitzner et al. 1981). On the Hanford Site, most nesting occurs in utility towers, on cliffs in Gable Butte, and in larger trees (Fitzner et al. 1981). Red-tailed hawks on the Hanford Site primarily feed on medium-sized mammals such as black-tailed jackrabbits and Townsend's ground squirrels, and on snakes (Fitzner et al. 1981).

A comprehensive survey of the mammals of the 200 Area plateau has not been conducted. Species known or likely to be present in the vicinity of the 200-BP-1 operable unit are listed in Table 3-12. Evaluations of rodents of the B-C cribs environment were conducted by Hedlund and Rogers (1976); blacktailed jackrabbit studies were conducted by Uresk et al. (1975); mule deer studies were conducted by Uresk and Uresk (1980). Two species of concern with regard to endangered species protection are Merriam's shrew, a candidate for listing by Washington State, and the Pacific western big-eared bat, a candidate for federal protection.

The most abundant small mammal on the 200 Area plateau is the Great Basin pocket mouse (O'Farrell 1975). Although primarily a granivore, the pocket mouse also consumes insects early in the year prior to seed production (Kritzman 1974). Pocket mice constitute the principal prey items in the diets of burrowing owls, great horned owls, long-eared owls, and barn owls foraging on the Hanford Site (Fitzner et al. 1981). Densities may range between 20 and 75 mice/ha in April depending on the habitat (Gano and Rickard 1982). Densities in cheatgrass habitat have been estimated at 30/ha (Hedlund et al. 1975). Burrows of this species were identified in the 200-BP-1 operable unit.

The second most abundant mammal on the Hanford Site is the deer mouse, which is nearly ubiquitous on the 200 Area plateau, though at much lower densities than pocket mice (Hedlund and Rogers 1976). Deer mice are herbivorous/granivorous, concentrating on green vegetation, especially tansy mustard and cheatgrass (Hedlund and Rogers 1976). Although nocturnal, they are found as occasional prey items in the diets of Swainson's hawks and red-tailed hawks nesting on the Hanford Site. More frequently, they are consumed by great horned owls, long-eared owls, burrowing owls, and barn owls (Fitzner et al. 1981).

Townsend's ground squirrels may also be abundant on the 200-Area plateau: Hedlund and Rickard (1981) identified Townsend's ground squirrels as the second most prevalent small mammal, with peak catches between one-half and one-fifth that for the Great Basin pocket mouse. Foraging preferences based on analyses of fecal samples identified Sandberg's bluegrass and tumble mustard as preferred food items (Rogers and Gano 1980). Townsend's ground squirrels are the principal food item for red-tailed hawks and the second most important item in the diet of post-fledgling Swainson's hawks fledged on the Hanford Site (Fitzner et al. 1981).

Black-tailed jackrabbits are found in nearly all habitats in the shrub-steppe region, and are the most common Lagomorph on the Hanford Site (Rickard et al. 1974). Black-tailed jackrabbits in the big sagebrush/Sandberg's bluegrass communities feed most heavily on needle-and-thread grass, yarrow, turpentine cymopterus, and tumble mustard (Uresk et al. 1975). Black-tailed hares are the principal prey of golden eagles wintering on the Hanford Site (Rickard et al. 1974), and are important constituents in the diets of great horned owls, long-eared owls, barn owls, ferruginous hawks, Swainson's hawks, and red-tailed hawks (Fitzner et al. 1981; Fitzner 1980).

Mule deer are common and widespread on the Hanford Site. Mule deer are usually dispersed throughout favorable habitats in small groups or singly. Mule deer occupying cheatgrass habitats near the 200 Areas are mainly forb and shrub consumers (Uresk and Uresk 1980). Natural mortality of mule deer fawns on the Hanford Site is relatively high, due in large part to coyote predation (Steigers and Flinders 1980). Mule deer home ranges on the site vary about a mean of approximately 40 km^2 (15 mi^2) (Eberhardt et al. 1982).

Coyotes are the most abundant carnivore on the Hanford Site. They have not been studied to any extent on the Hanford Site except on the Arid Lands Ecology Reserve. Their diet is diverse, reflecting the availability of prey. Where the Great Basin pocket mouse is most abundant in the habitat, they have been found to be most abundant in the coyote diet (Stoel 1976). Other prey include leporids, voles, pocket gophers, ground squirrels, mule deer fawns, birds, reptiles, beetles, and grasshoppers (Stoel 1976, Steigers and Flinders 1980).

Badgers are a common mammalian carnivore on the Hanford Site. Badger excavations were found in the nonradiation area of the 200-BP-1 operable unit (1989 biolocial survey). Badger densities are unknown on the site. Estimates of badger densities in similar habitats of southcentral Idaho ranged from 2-7 badgers per km^2 (Messick and Hornocker 1981). Badger home ranges cover approximately 0.2 - 4.0 km^2 (Messick and Hornocker 1981). Badgers feed primarily on ground squirrels, small mammals, and arthropods (Messick and Hornocker 1981).

Reptiles

An assessment of reptiles at the 200-BP-1 operable unit has not been performed. Reptiles that could occur near the 200-BP-1 operable unit include the western yellow-bellied racer (*Coluber constrictor*), the Great Basin gopher snake (*Pituophis melanoleucus*), the northern Pacific rattlesnake (*Crotalus viridus*), the desert nightsnake (*Hypsiglena torguata*), the striped whipsnake (*Masticophis taeniatus*), the sagebrush lizard (*Sceloporus graciosus*), the side-blotched lizard (*Utastansburiana*), and the pygmy short-horned lizard (Phrynosoma douglassi) (Rogers and Rickard 1977). The most common reptiles found in the big sagebrush/Sandberg's bluegrass habitat of the 200 Area plateau are side-blotched lizards and yellow-bellied racers (Marr et al. 1988). Side-blotched lizards were found in approximate densities of 15 lizards/ha in the B-C cribs area in the 1970's (Rogers and Rickard 1977). There are no Federal or State-classified threatened, endangered, or candidate species among the reptiles of Hanford. Some reptiles, principally the western yellow-bellied racer and the gopher snake, are major prey for regulated raptors such as Swainson's and red-tailed hawks on the Hanford Site.

Insects

Insects and spiders are an important component of the plant communities of the 200 Area plateau in terms of biomass and ecological role. Invertebrate densities in sagebrush/bunchgrass habitat on Arid Lands Ecology Reserve ranged from 450 to nearly 2000 individuals/m^2, with a biomass of up to 0.5 g/m^2 (Rogers 1977). The predominant taxa include ground-dwelling darkling beetles (family Tenebrionidae), and shrub-dwelling bugs (order Homoptera), grasshoppers (order Orthoptera), true bugs (order Hemiptera), and spiders (order Araneida) (Rogers 1979). No insects of the Hanford Site are listed by the State or Federal governments as threatened, endangered, or candidates for listing. The survey of the 200-BP-1 operable unit (1989) noted the presence of a number of harvester

ant (Pogonomyrmex spp.) mounds within the surface contamination area. These ants may serve as a significant means of transporting subsurface waste to the surface.

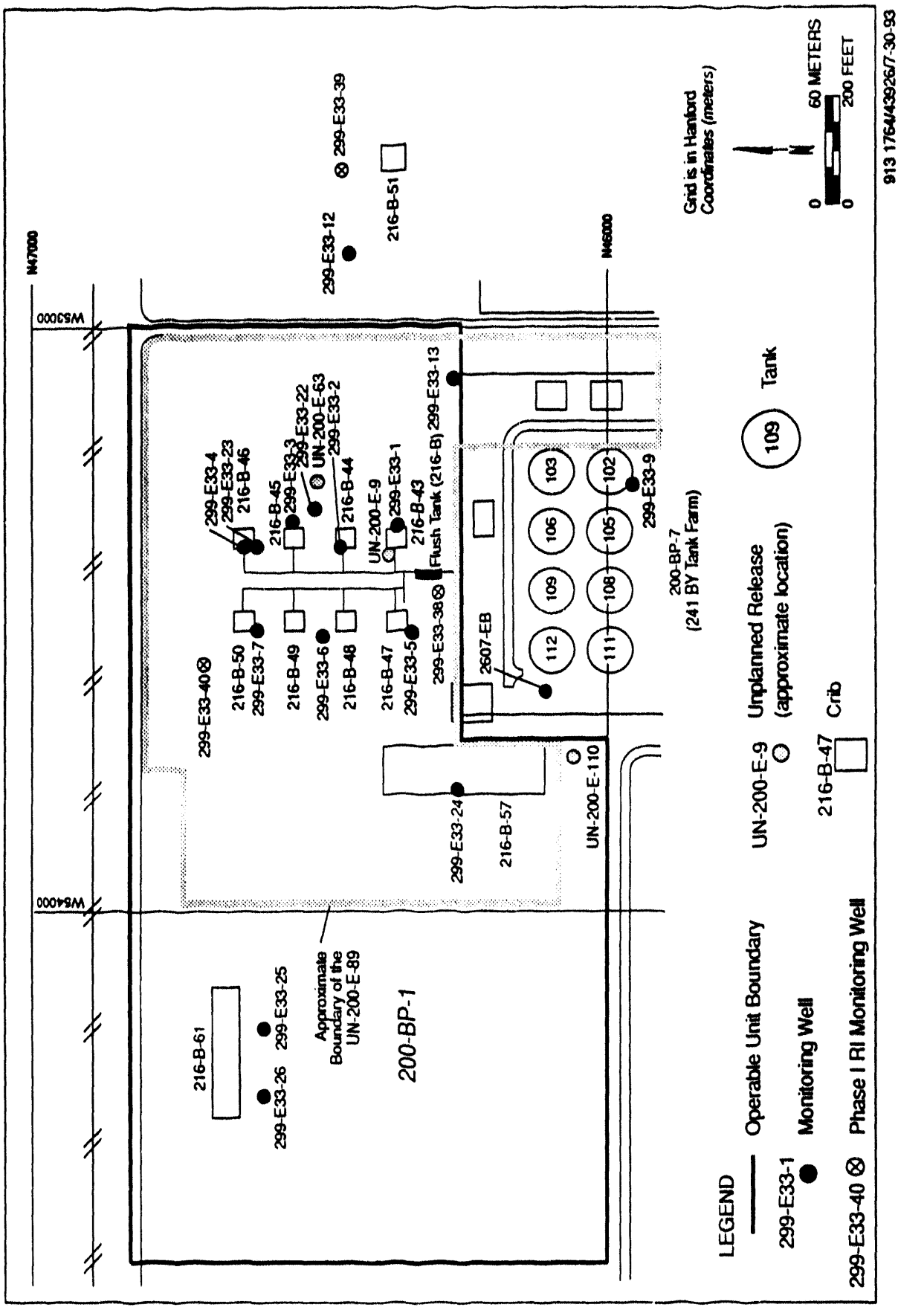
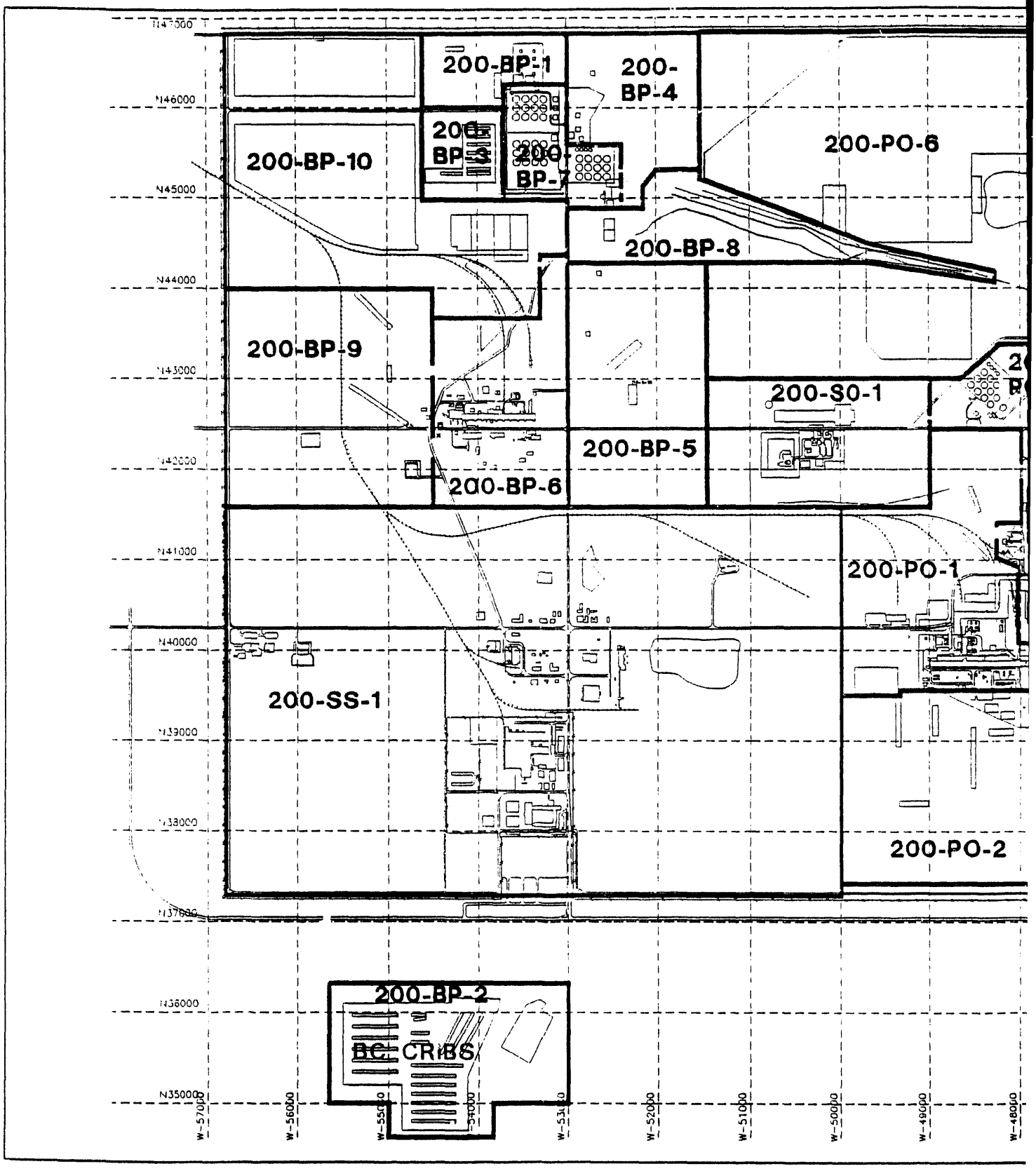


Figure 3-1. Site Map of the 200-BP-1 Operable Unit.


200-BP-1
200-BP-4
200-BP-10
200-PO-6
200-BP-8
200-BP-9
200-SO-1
200-BP-5
200-BP-6
200-PO-1
200-SS-1
200-PO-2
200-BP-2
BC CRIBS
N46000
N45000
N44000
N43000
N41000
N40000
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N38000
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W-52000
W-51000
W-50000
W-49000
W-48000

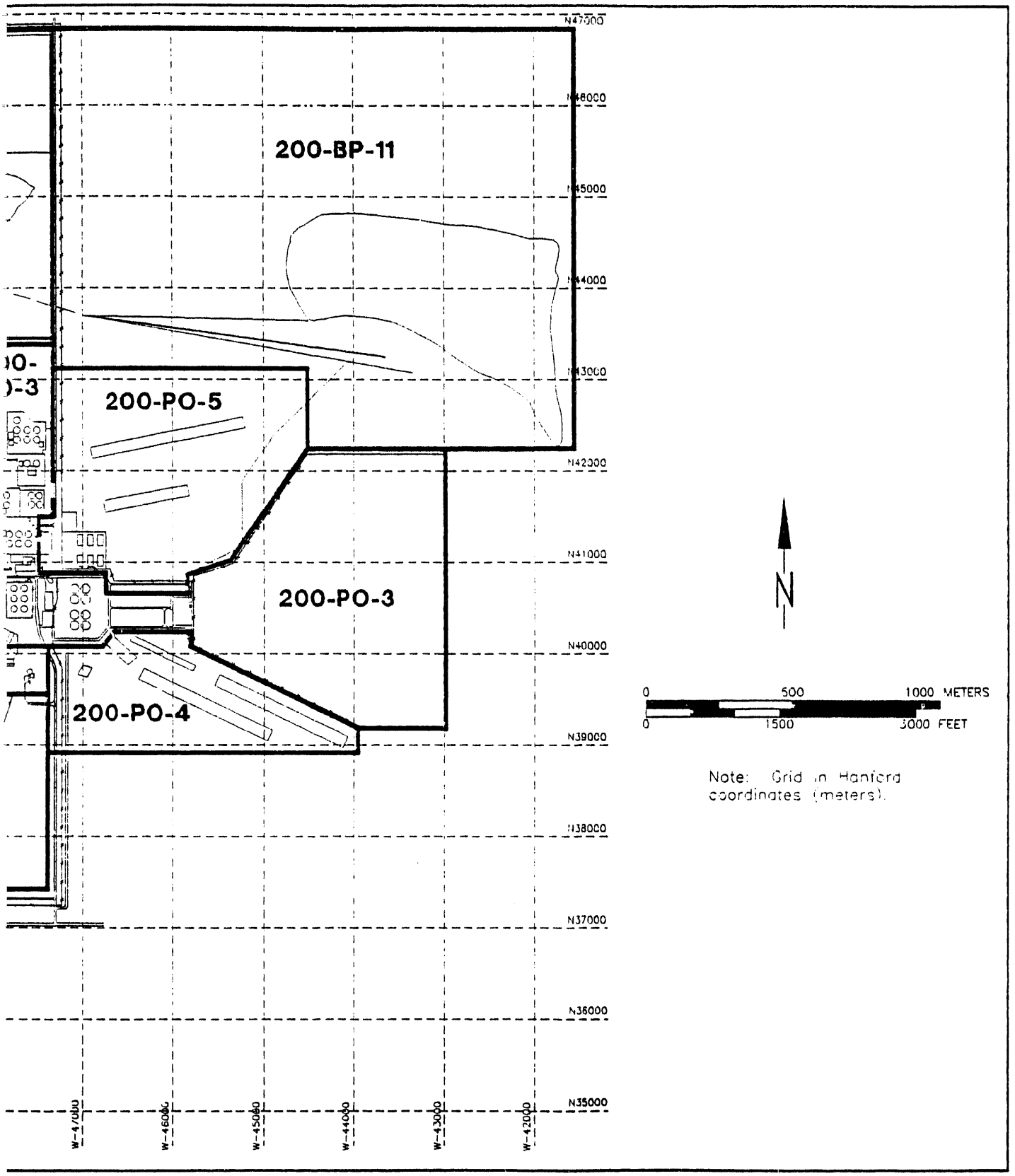

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Figure 3-2. 200 East Area Operable Units.

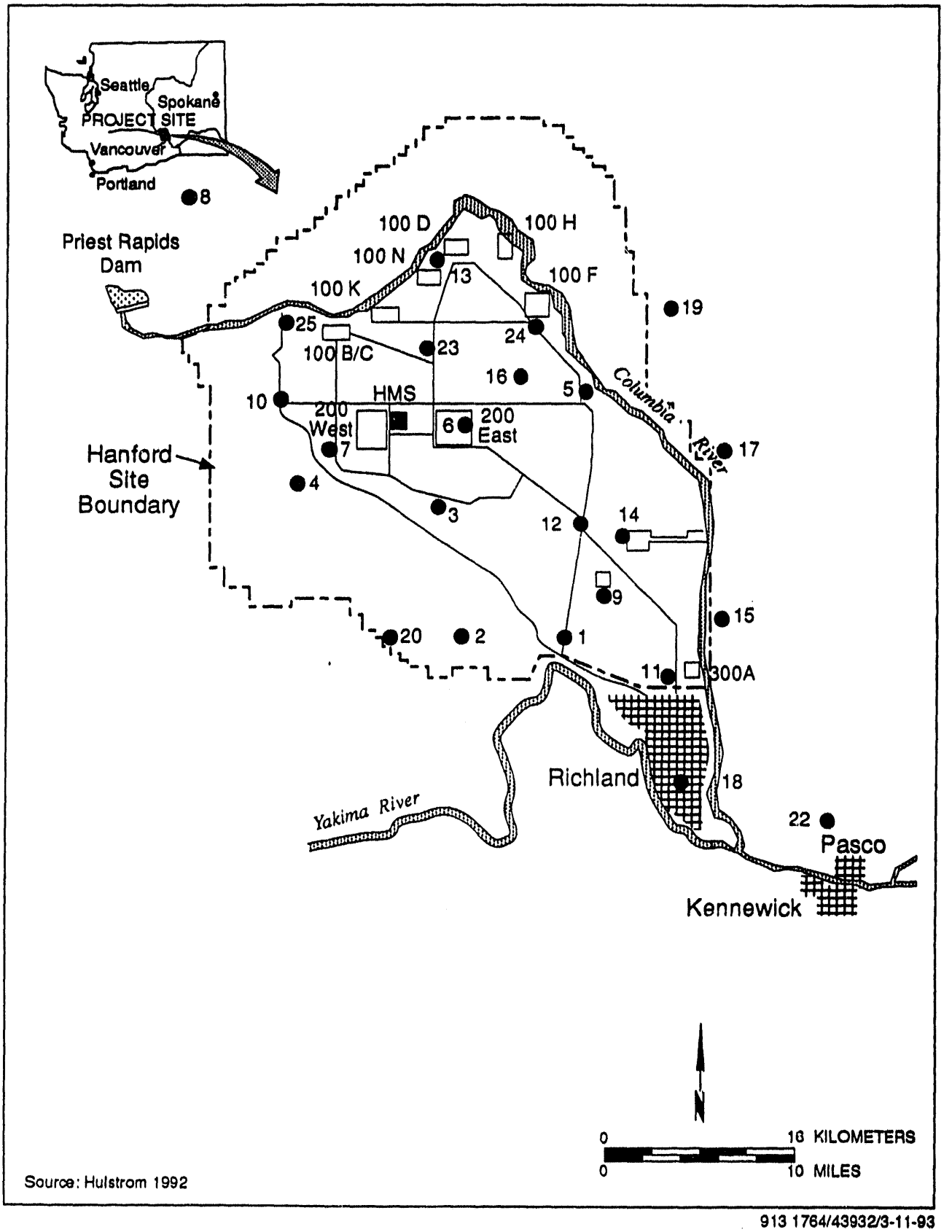

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Figure 3-3. Meteorologic Monitoring Stations at the Hanford Site.

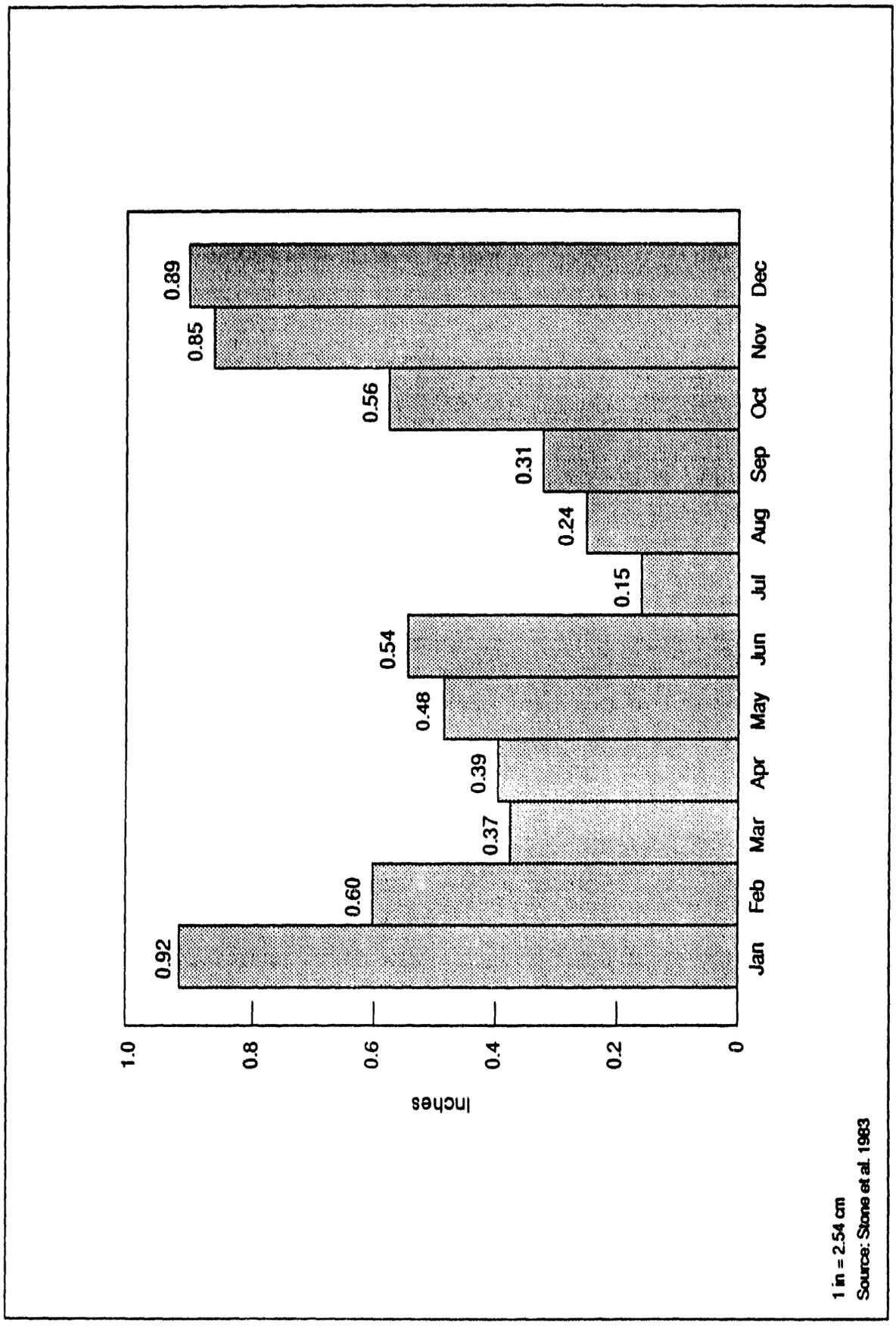
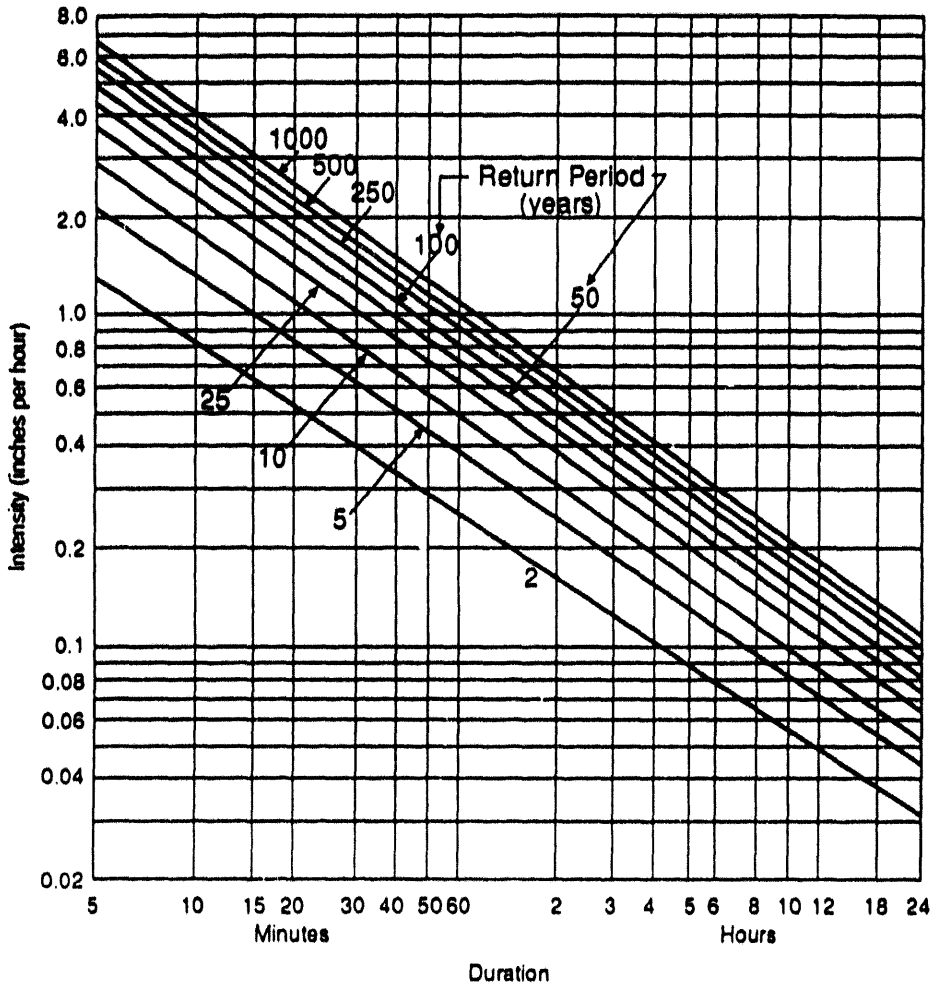


Figure 3-4. Monthly Average Precipitation Amounts, 1912 through 1980.



Source: Stone et al. 1983.

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Figure 3-5. Rainfall Intensity Duration and Frequency Based on the Period 1947 to 1969 at Hanford.

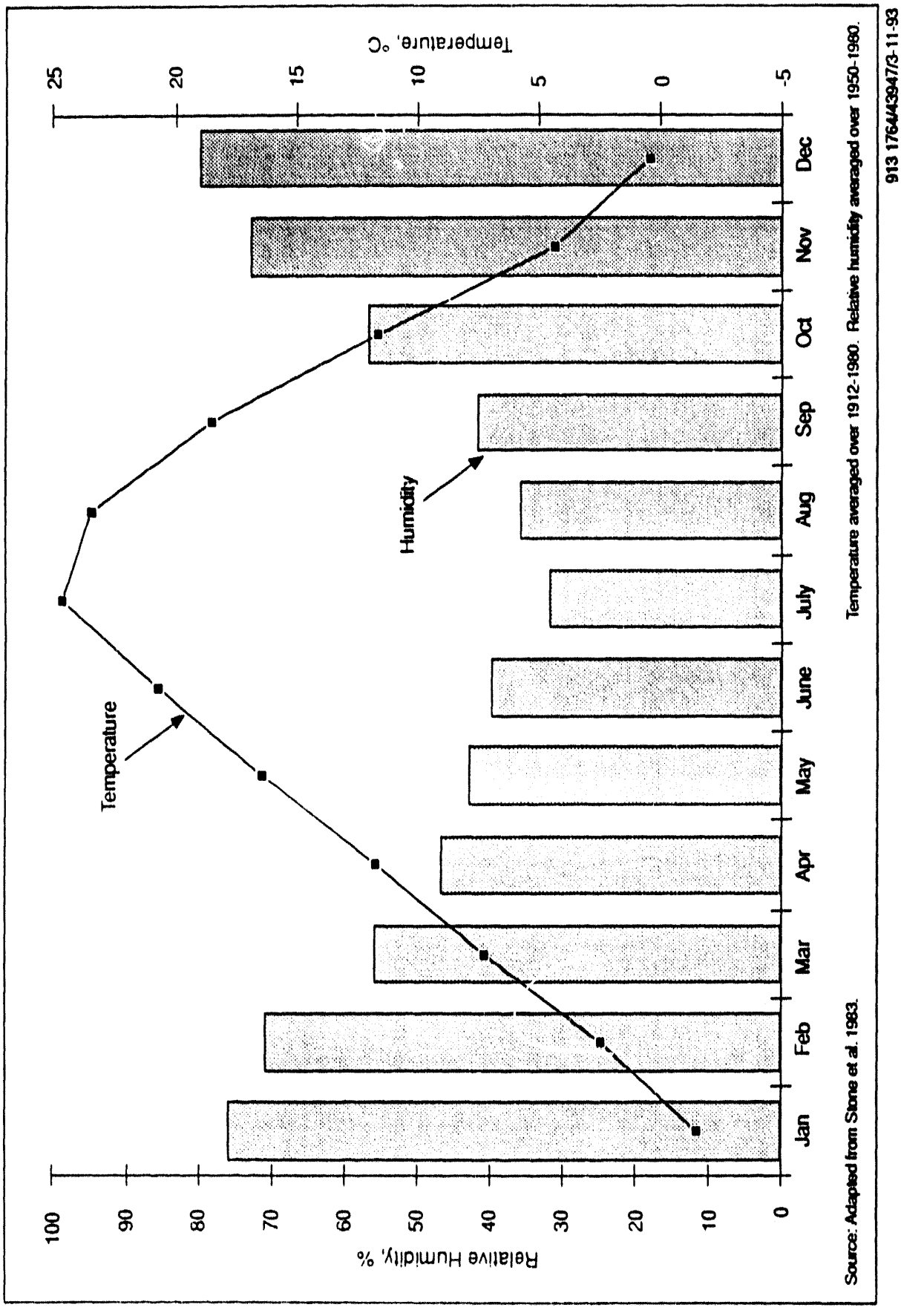

Figure 3-6. Average Monthly Temperature and Relative Humidity at the HMS.

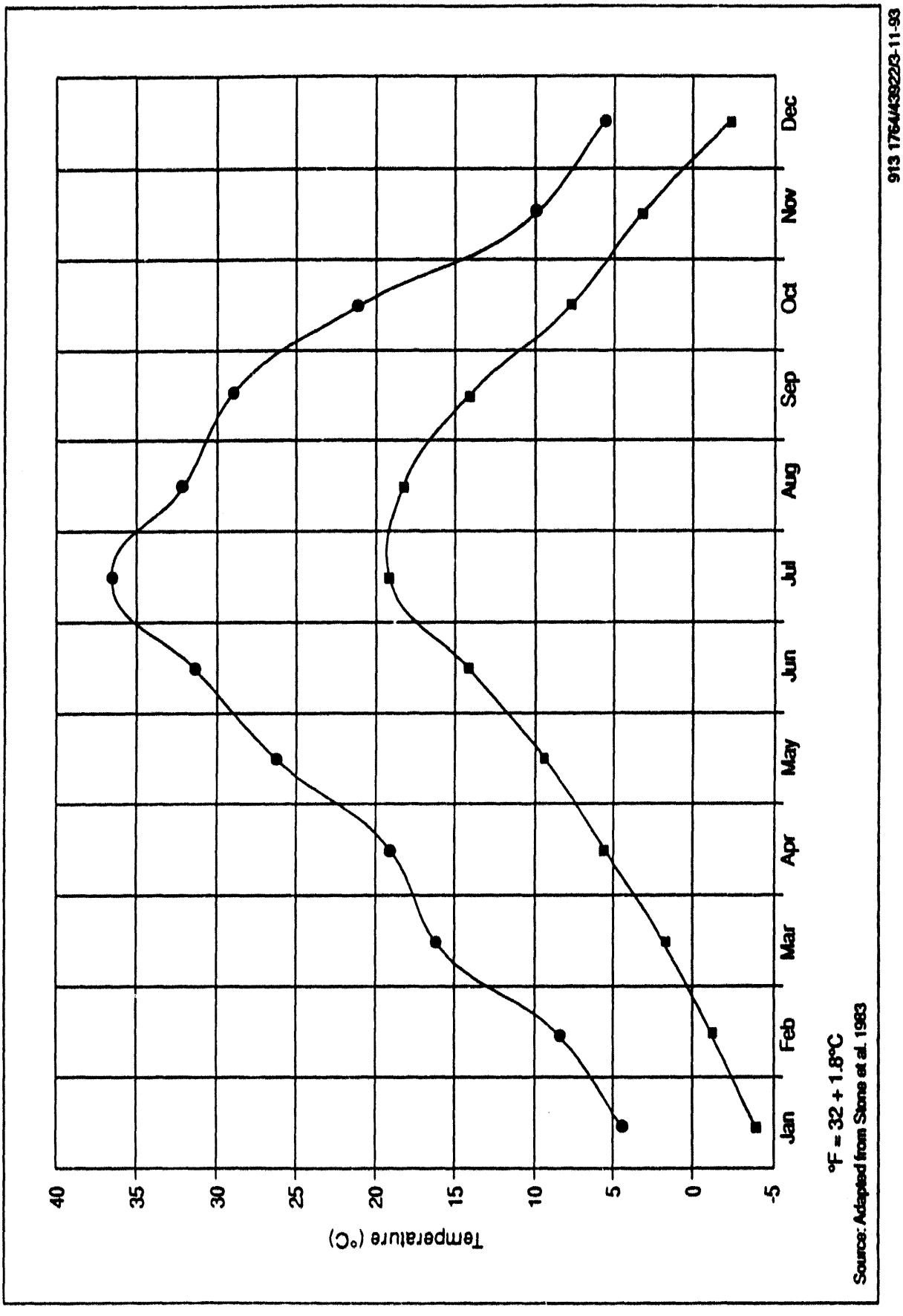

Figure 3-7. HMS Monthly Average High and Low Air Temperatures, 1951 through 1980.

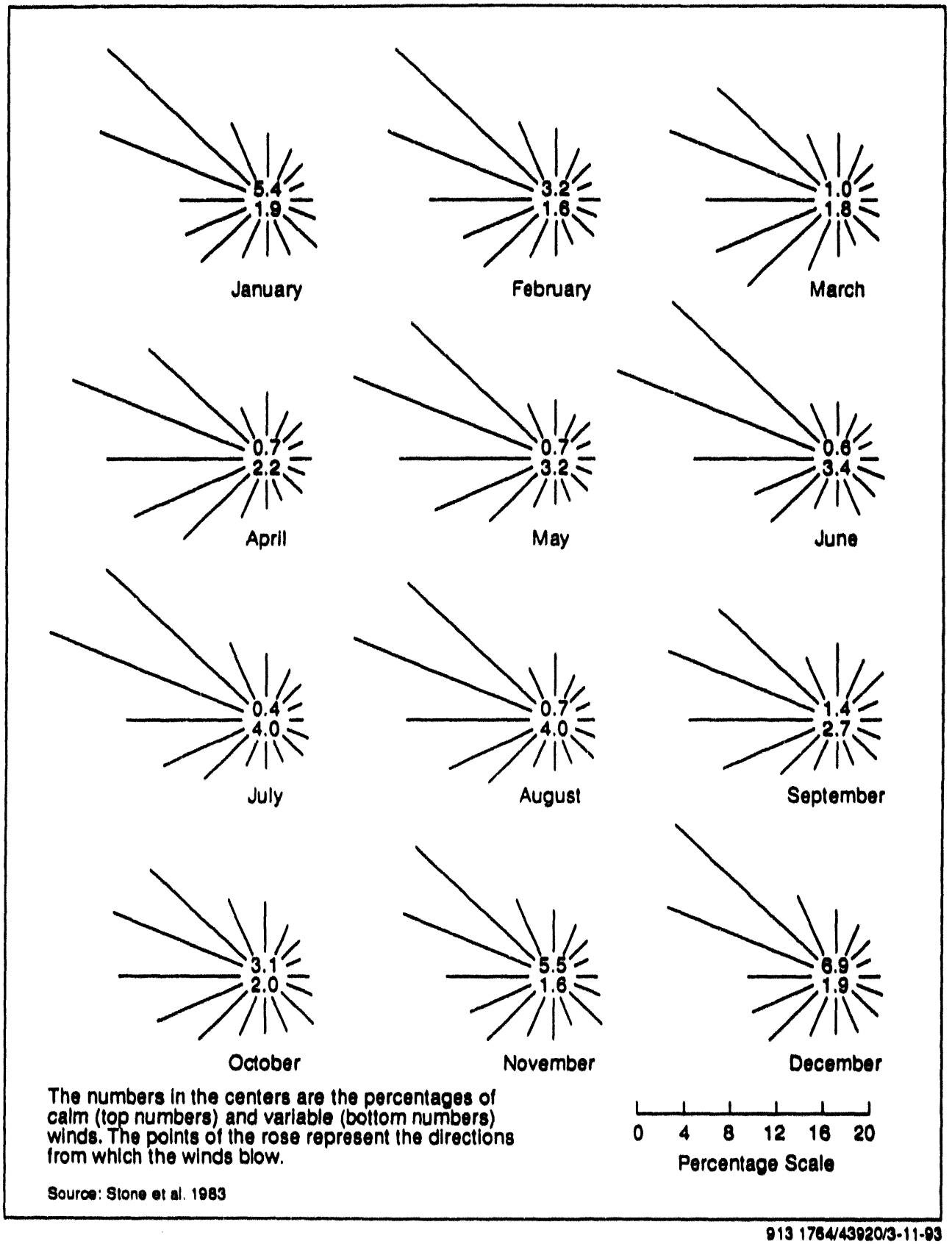

Figure 3-8. Monthly Wind Roses for HMS Based on 50 foot Wind Data, 1955 through 1980.

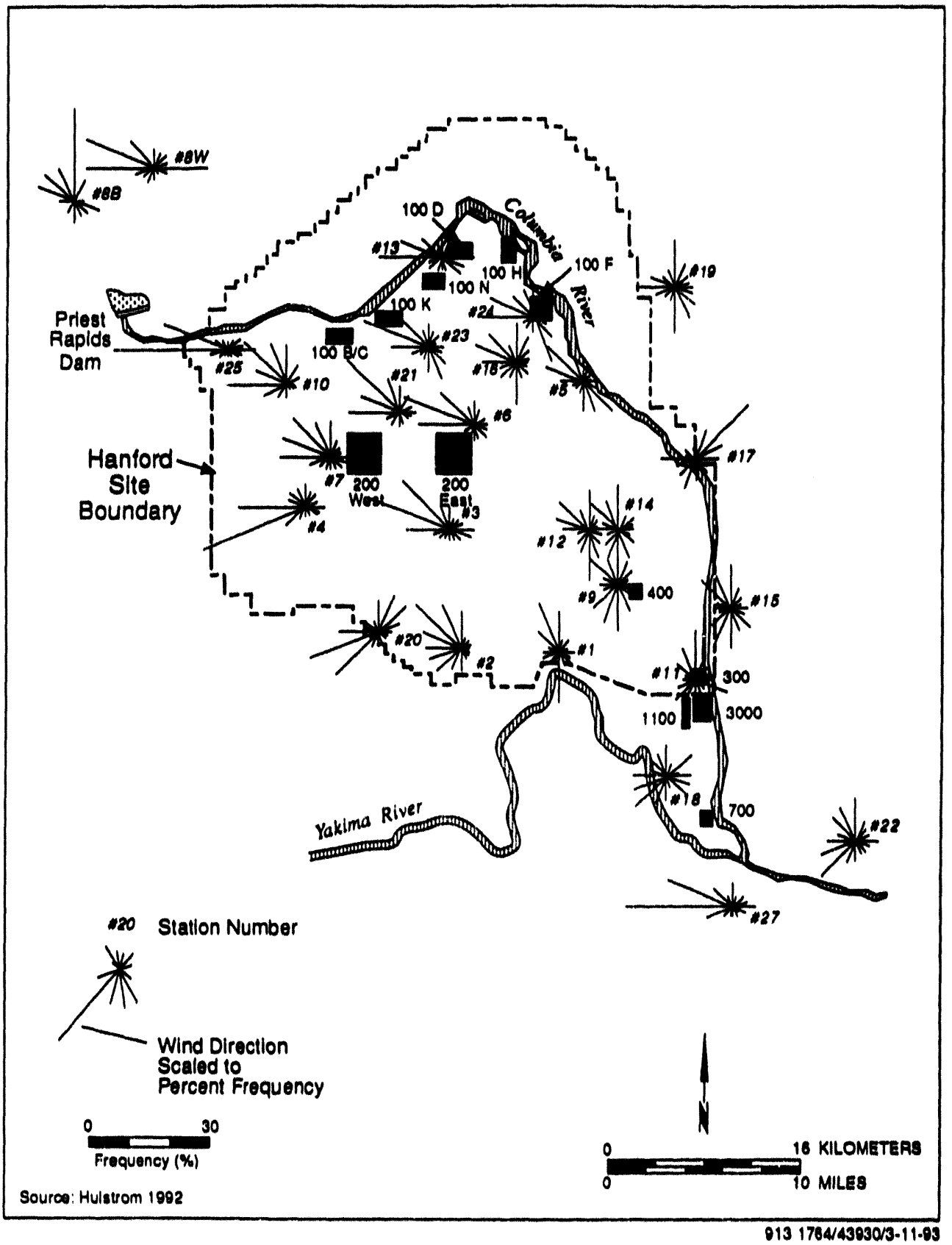

Figure 3-9. Wind Roses for the Hanford Site.

Figure 3-10. Average Monthly Pan Evaporation at Prosser, WA for the Period 1924-1988. (Measurements not taken for November through March)

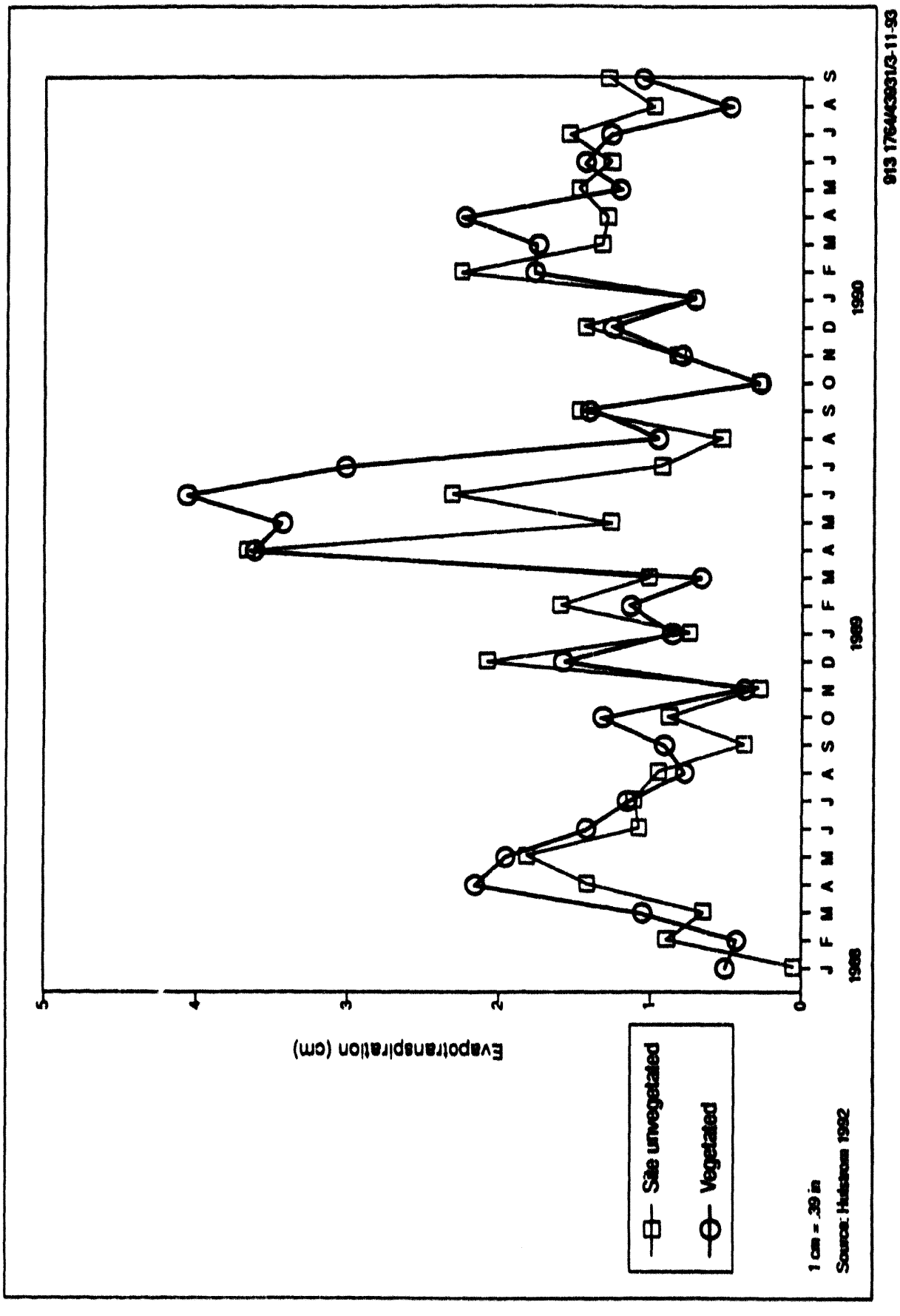

Figure 3-11. Total Monthly Evapotranspiration Near the 300 Area.

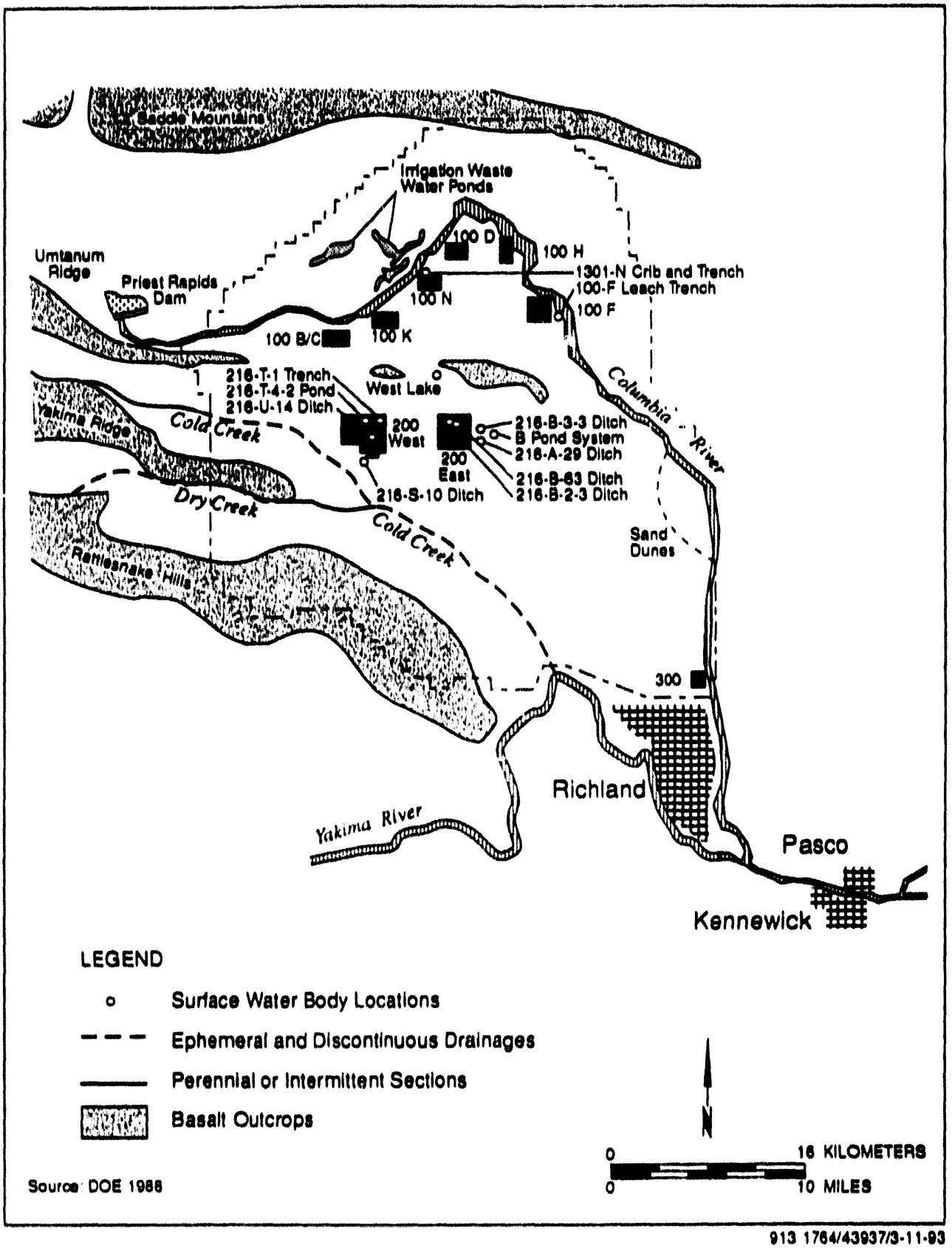

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Figure 3-12. Surface Water Bodies, Including Ephemeral Creeks, on the Hanford Site.

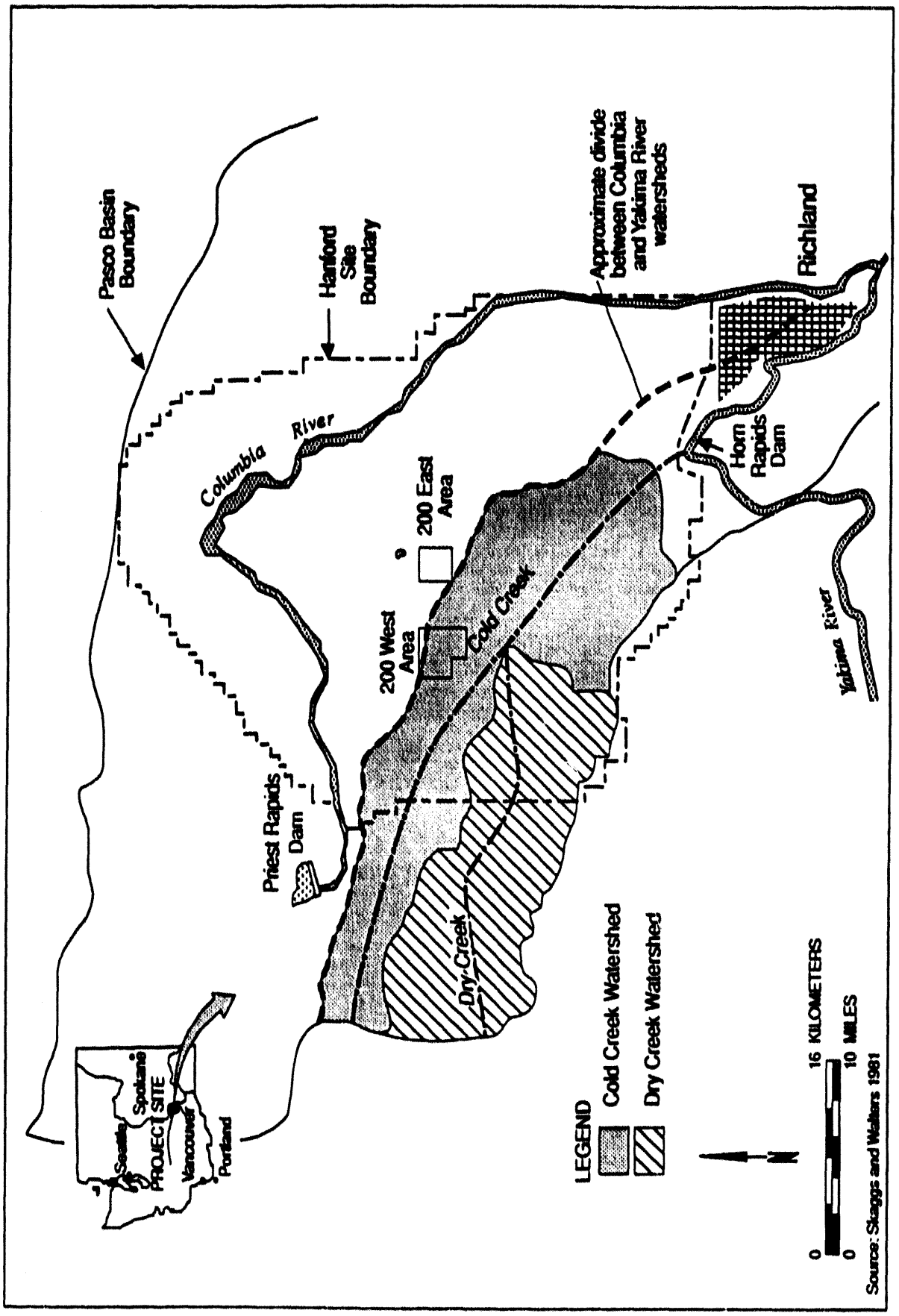

Figure 3-13. Approximate Location of Major Drainage Divides.

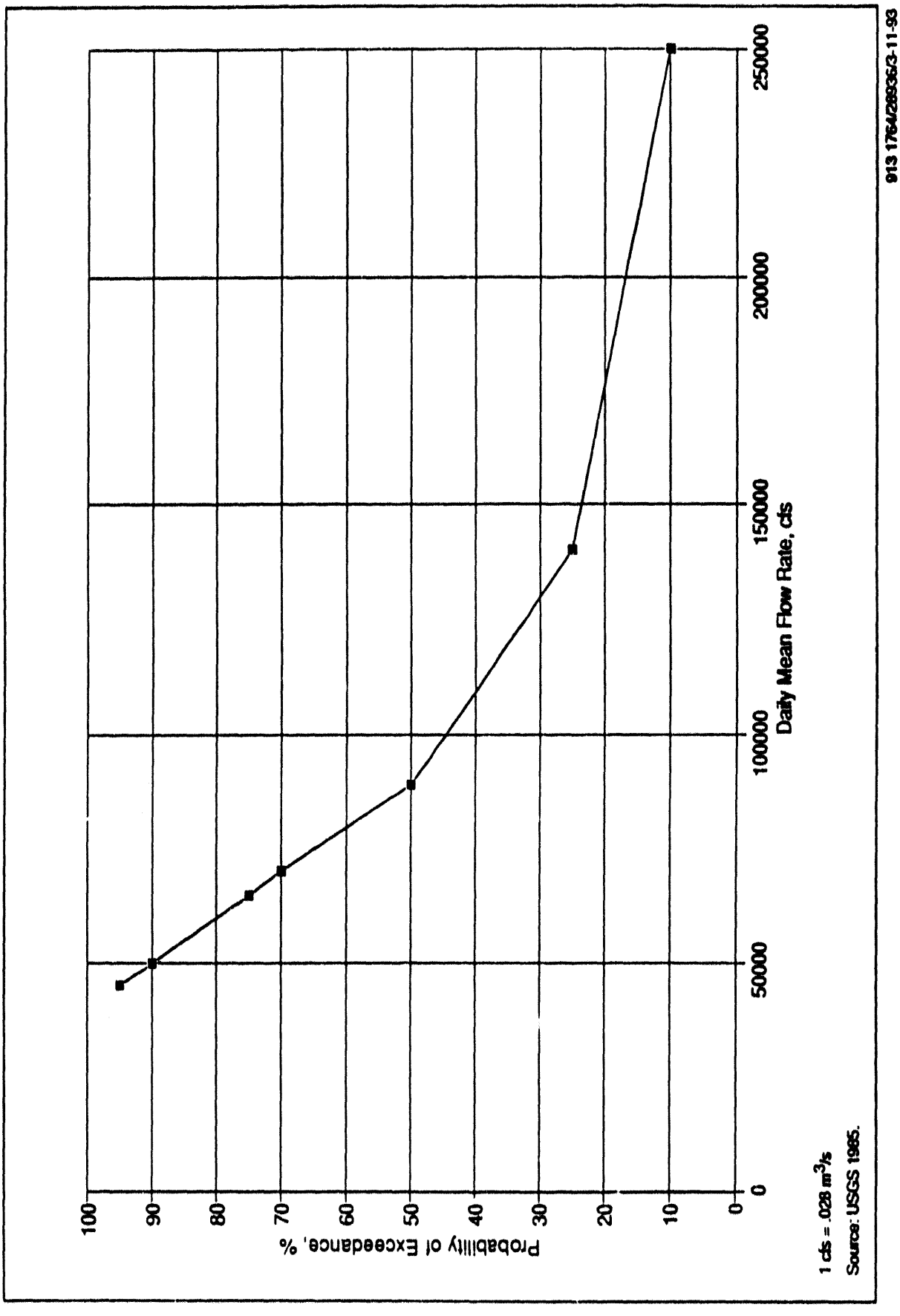

Figure 3-14. Columbia River Flow Duration Curve for Priest Rapids Dam. (Based on data collected over the period 1939-1979)

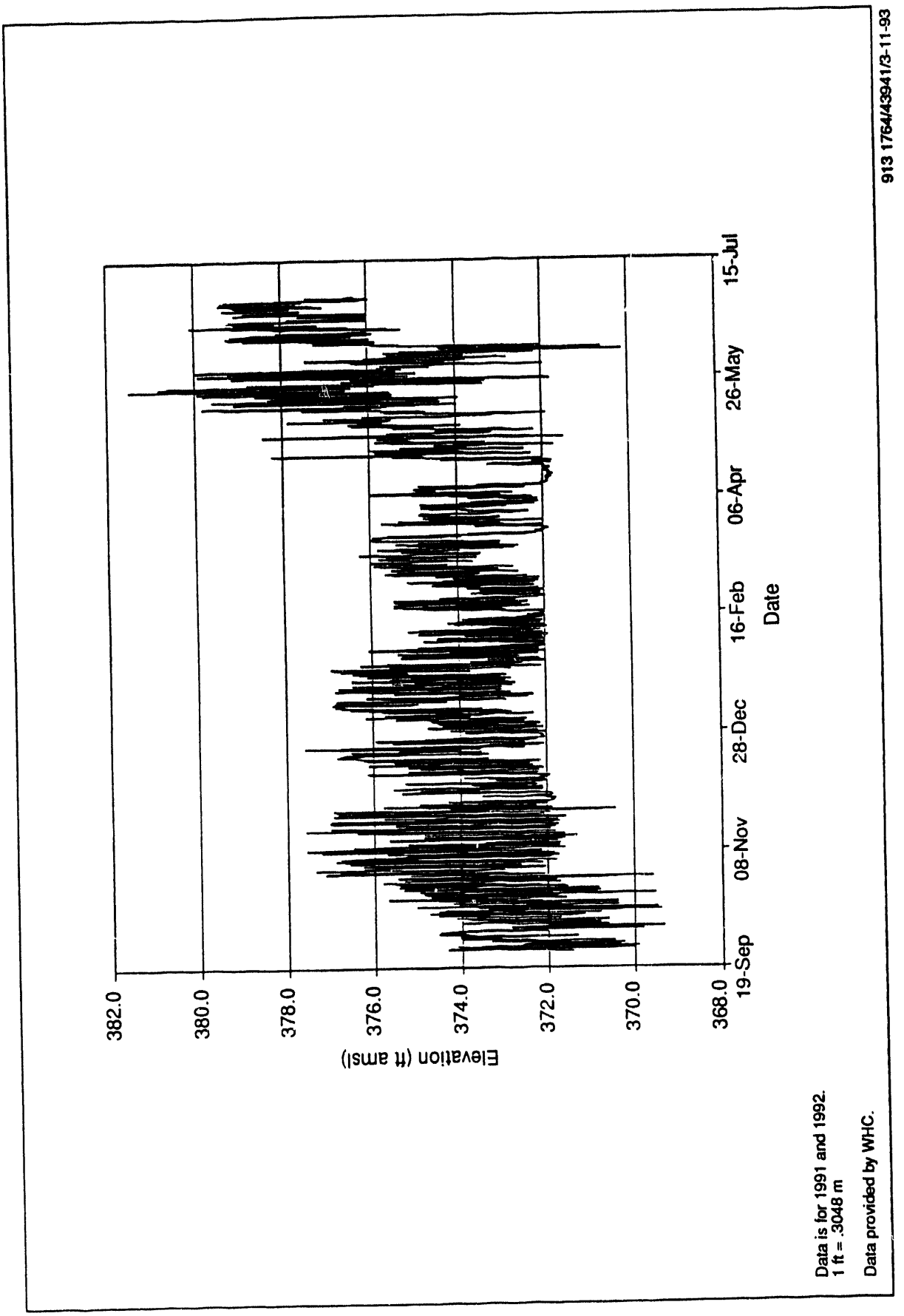

Figure 3-15. Columbia River Level Variations at the 100-H Area.

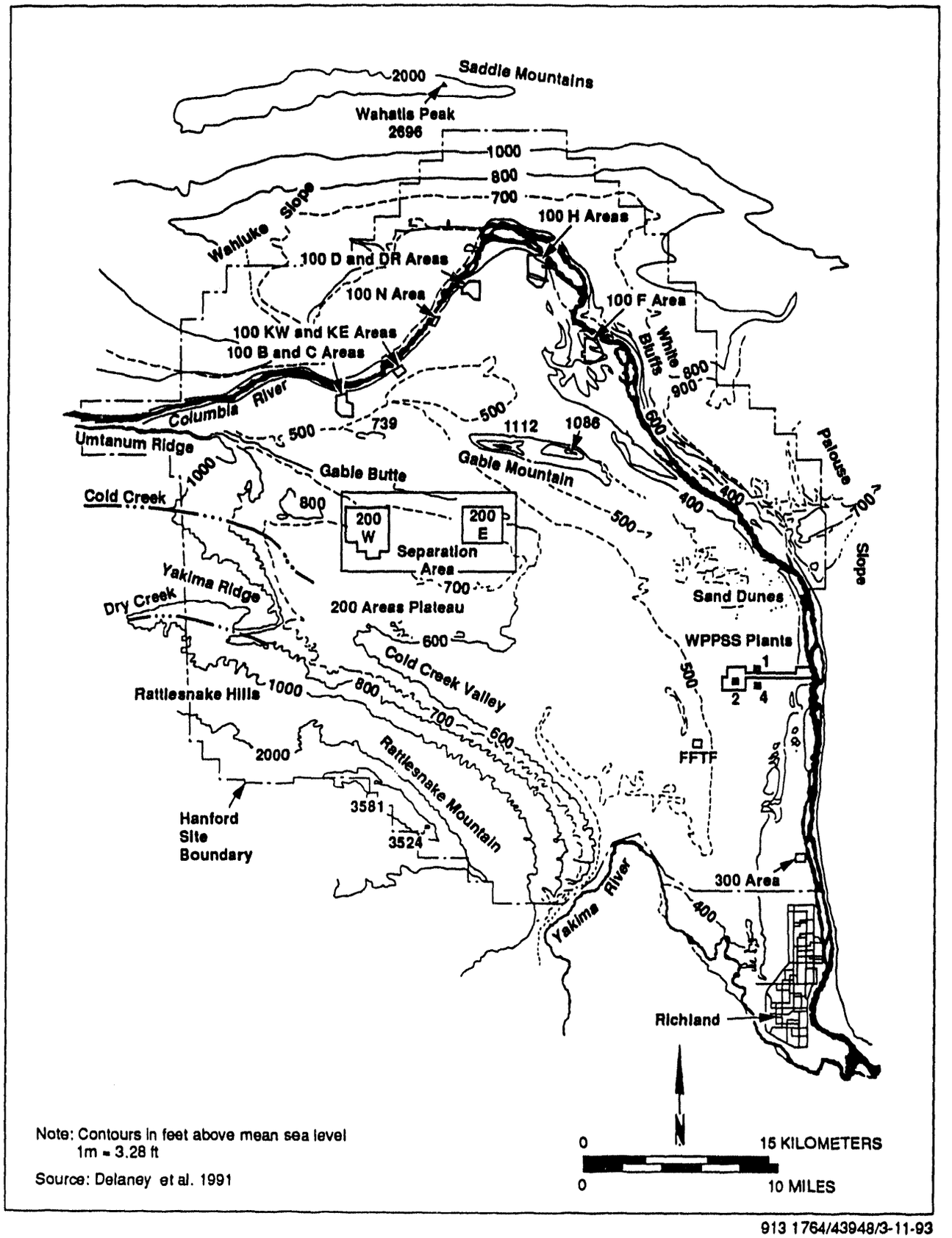

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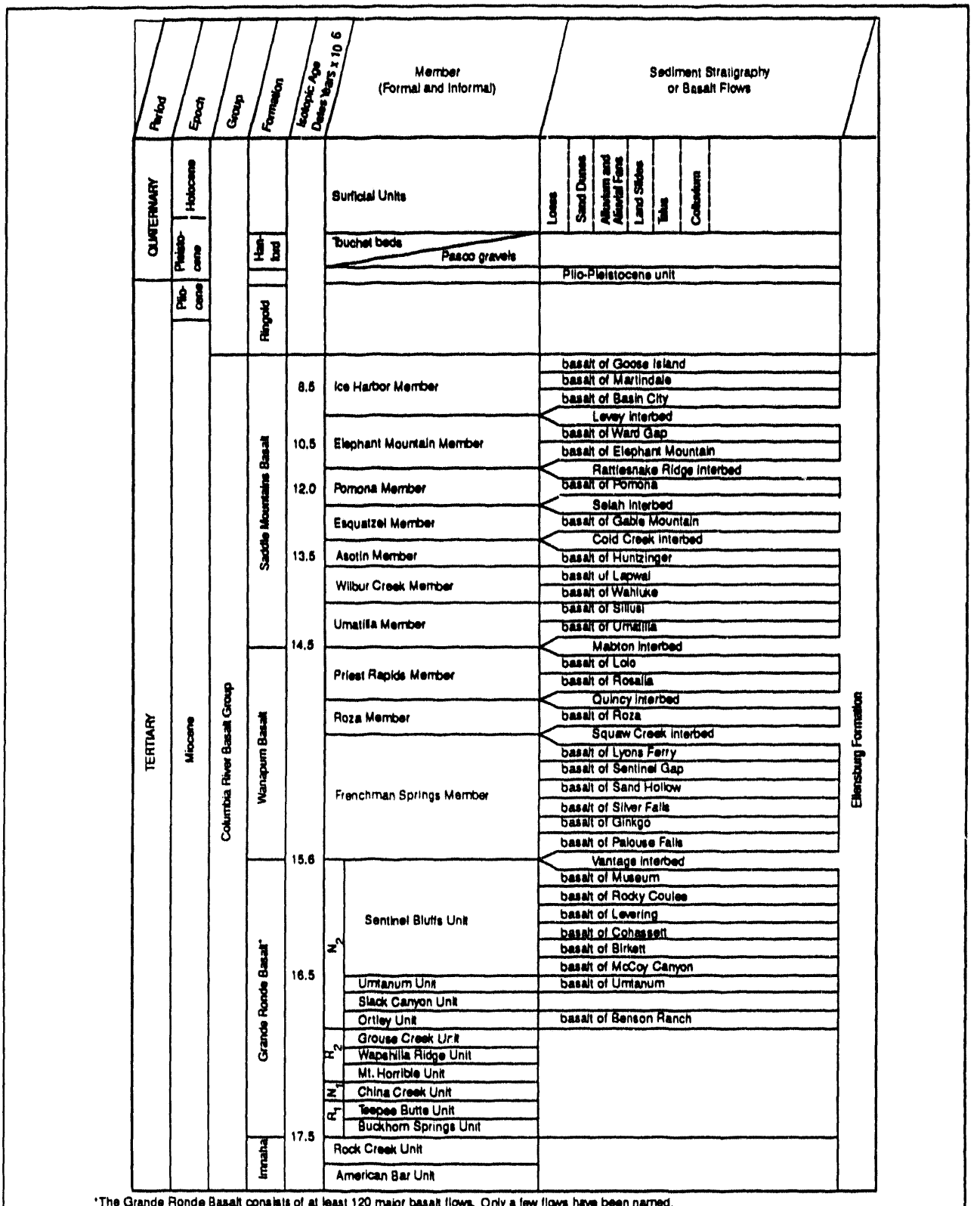
Figure 3-16. Location and Topography of the Hanford Site.



*The Grande Ronde Basalt consists of at least 120 major basalt flows. Only a few flows have been named.
N_2, R_2, N_1 and R_1 are magnetostratigraphic units.

Source: Delaney et al. 1991

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Figure 3-17. Basalt and Suprabasalt Stratigraphy at the Hanford Site, Washington.

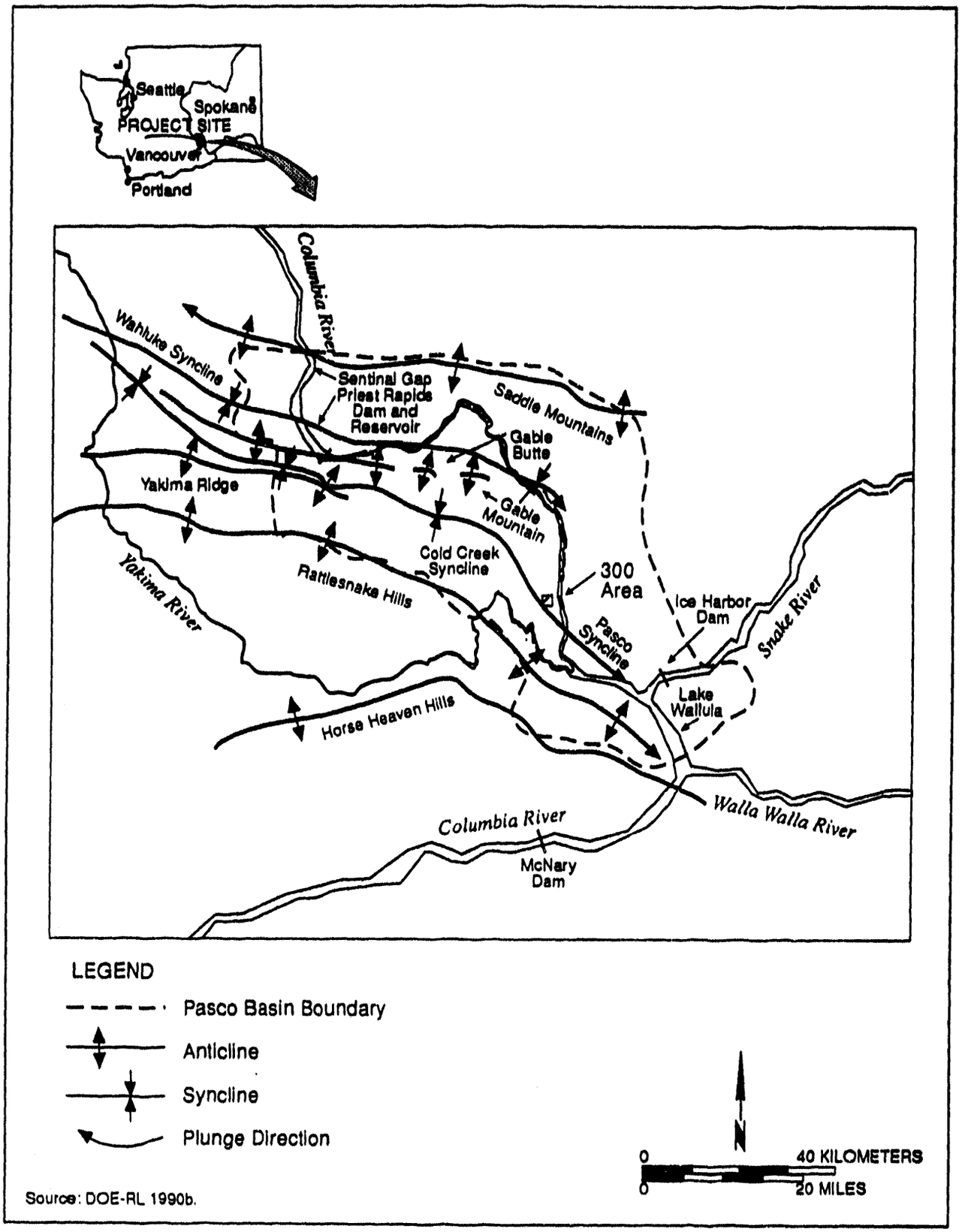
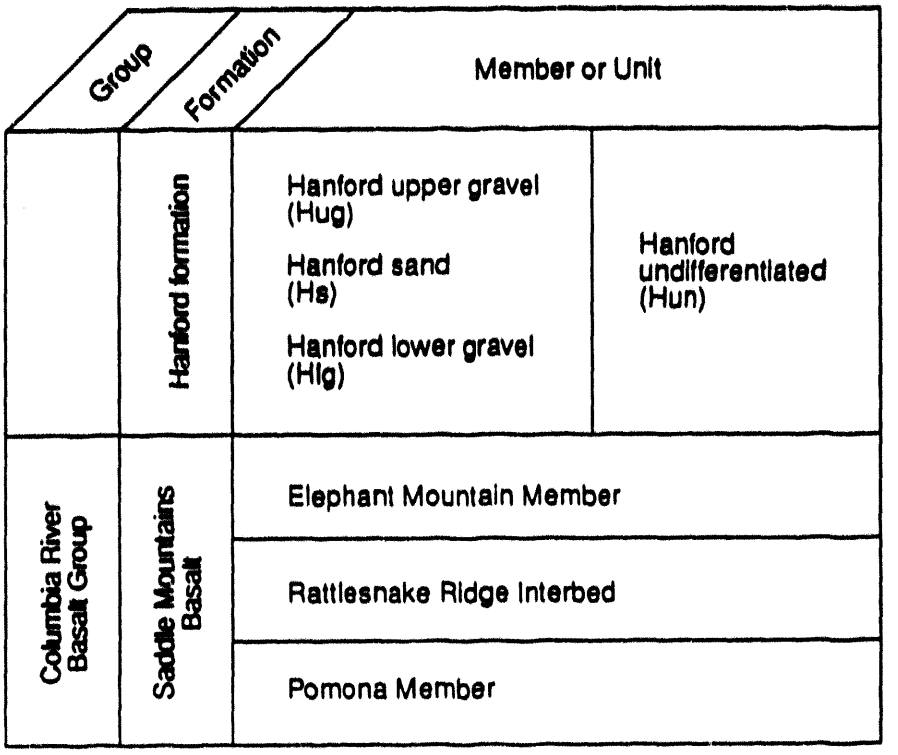


Figure 3-18. Major Folds in the Pasco Basin.



Source: Hoffman et al. 1992.

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Figure 3-19. Generalized Stratigraphic Column of the 200-BP-1 Operable Unit.

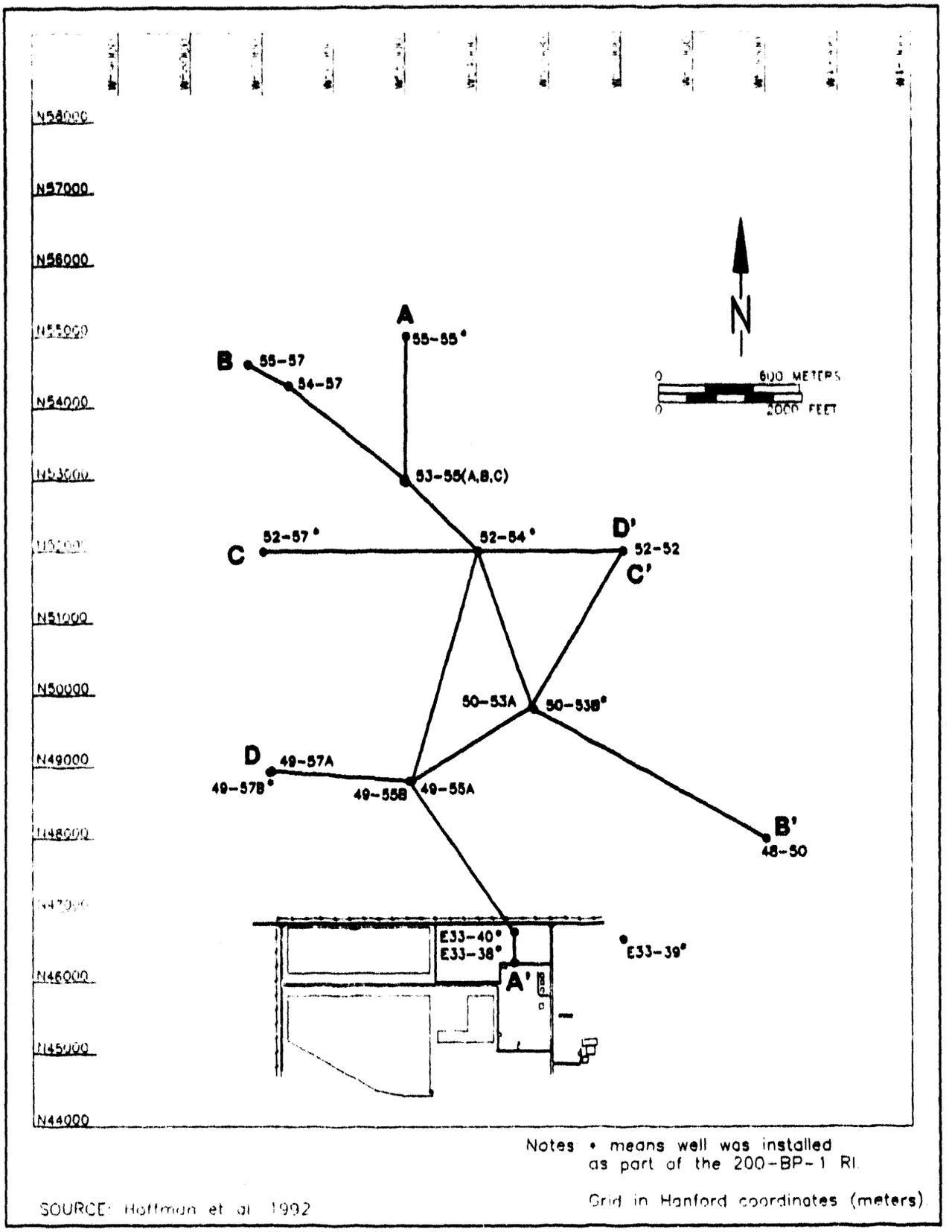


Figure 3-20a. 200-BP-1 Operable Unit Map Cross Section Locations.

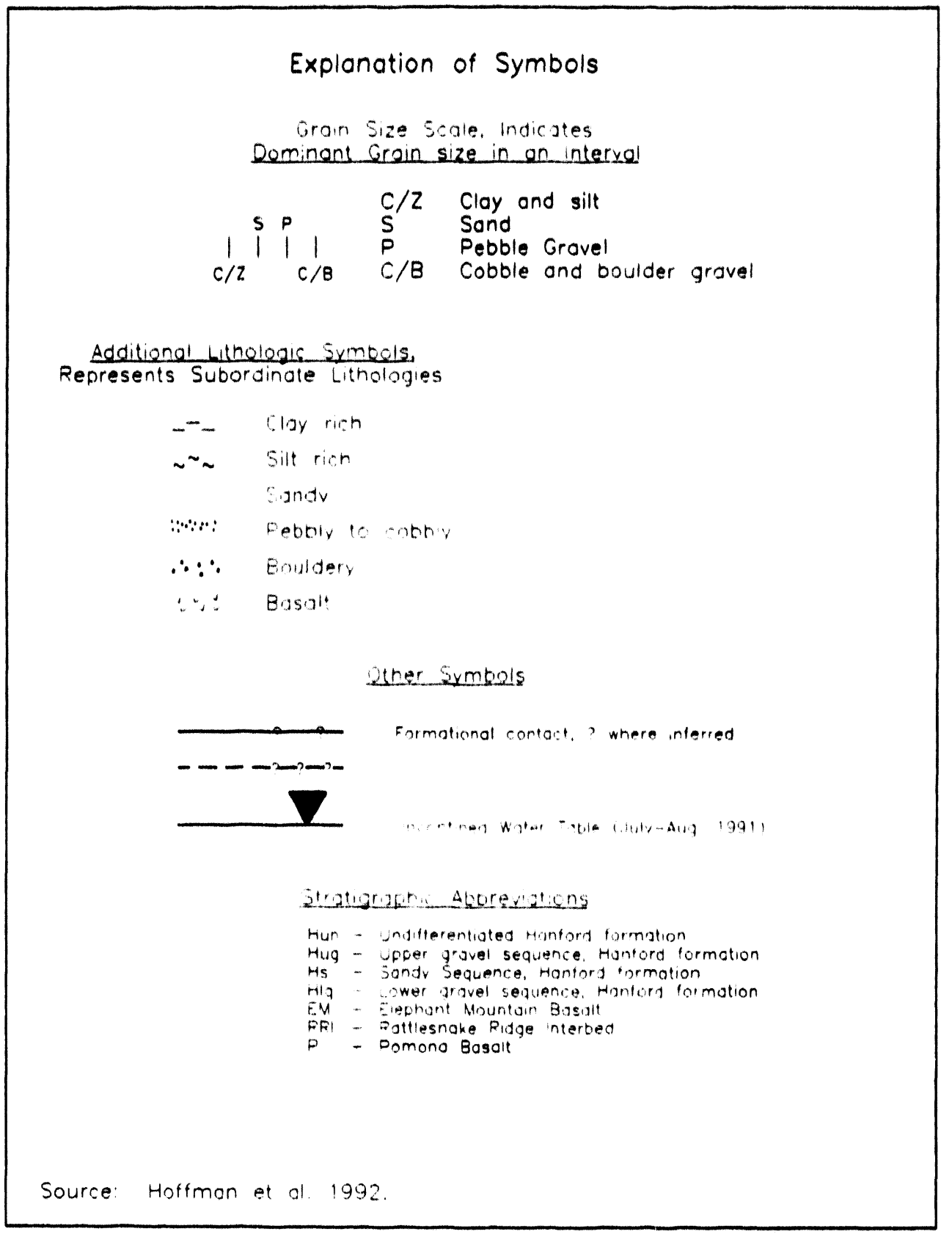
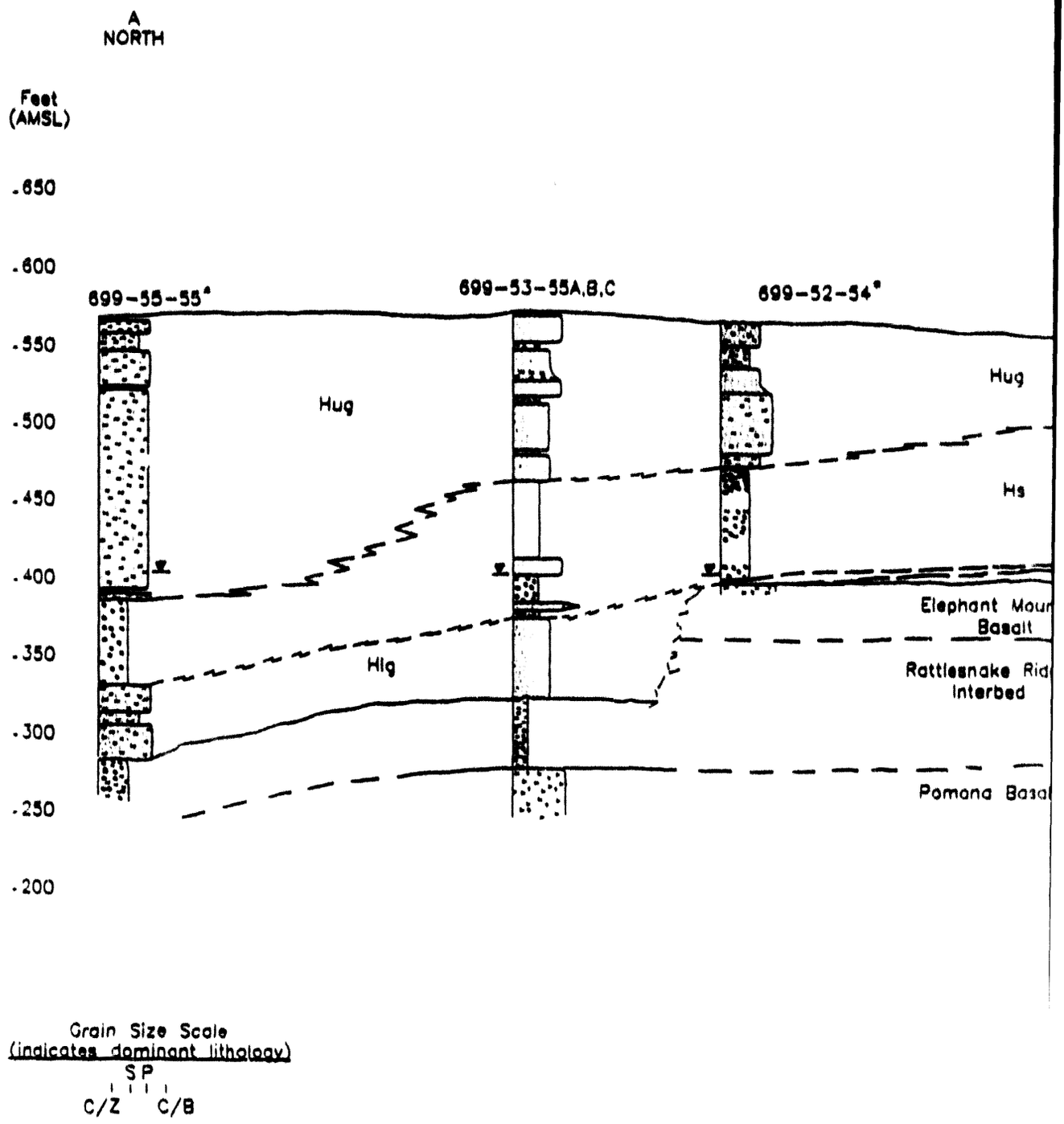


Figure 3-20b. Explanation of Symbols Used in Geologic Cross-Sections.



Source: Modified from Hoffman et al. 1992.

Note: * means well installed as part of the 200-BI

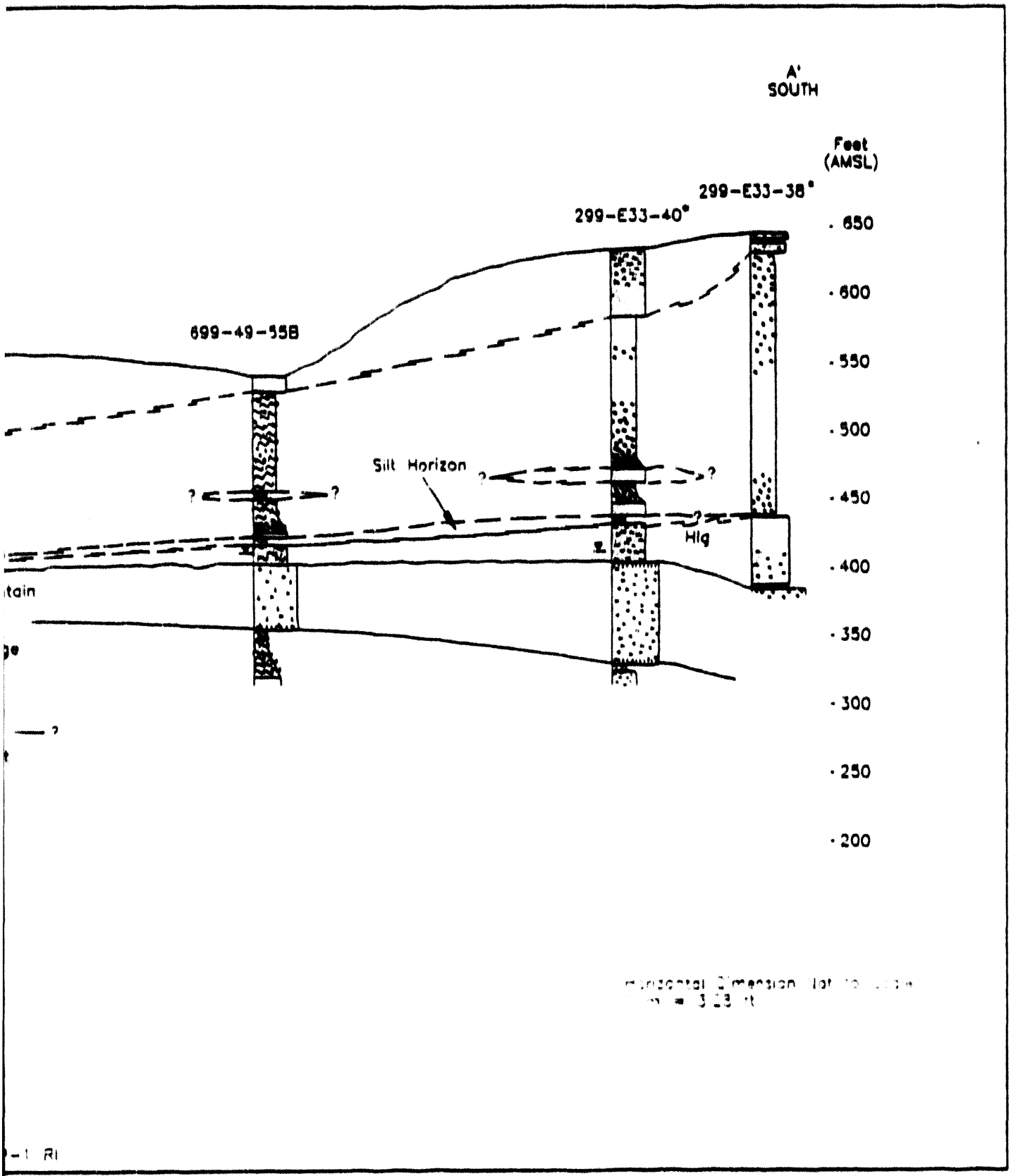
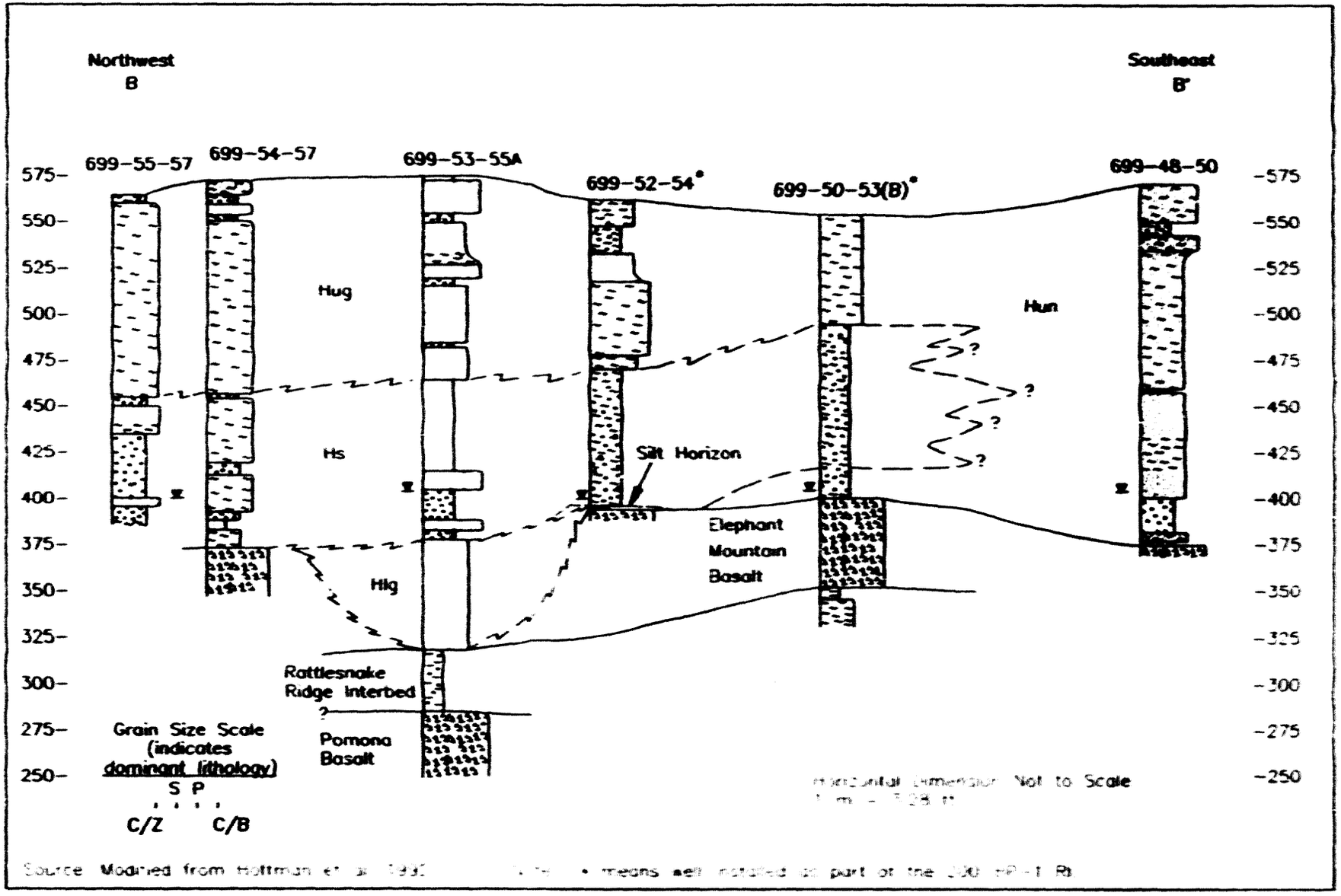


Figure 3-21. Cross Section A-A'.



Source Modified from Hoffman et al. 1992. * means well installed in part of the 200 East Area

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Figure 3-22. Cross Section B-B'

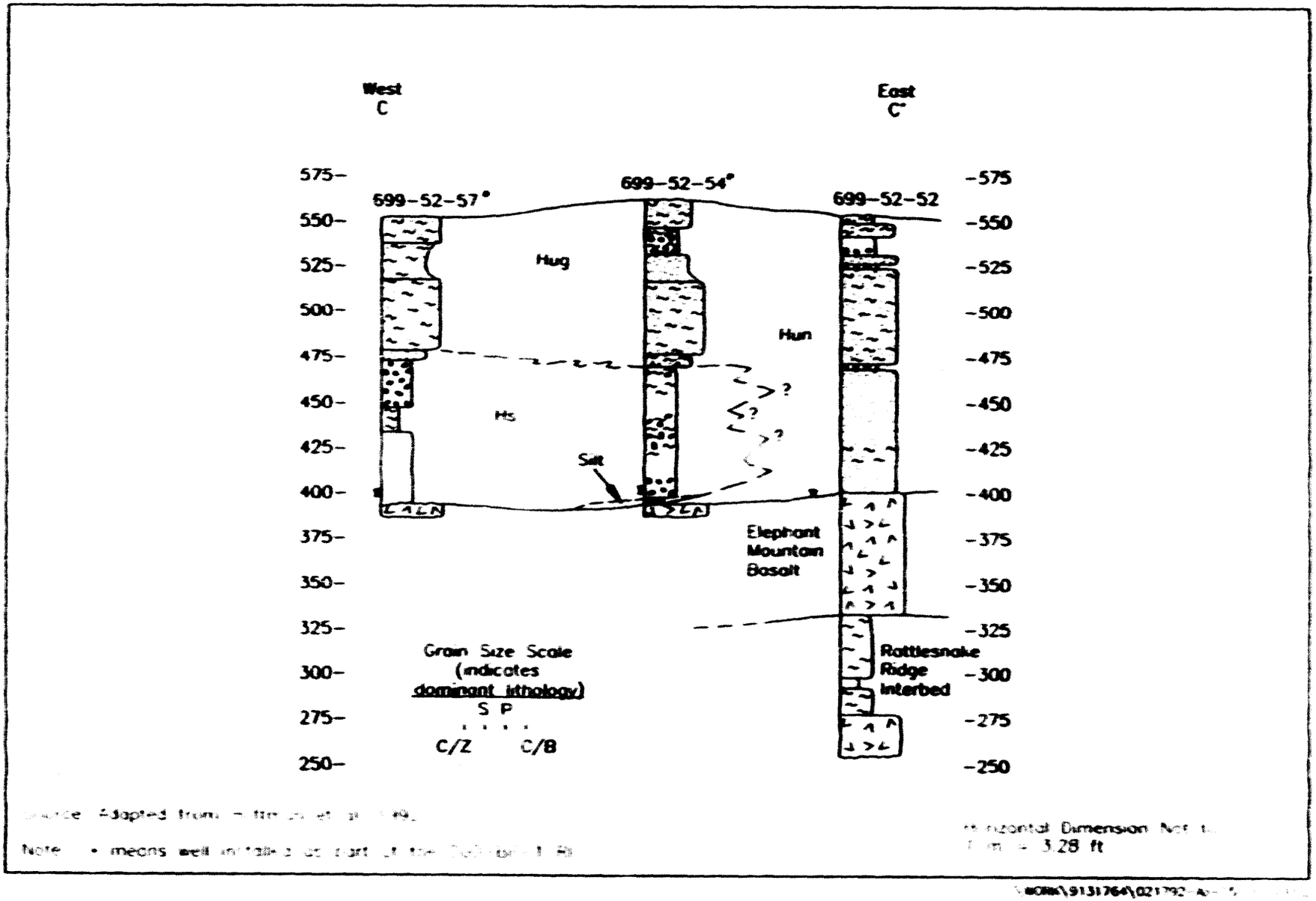

Figure 3-23. Cross Section C-C'.

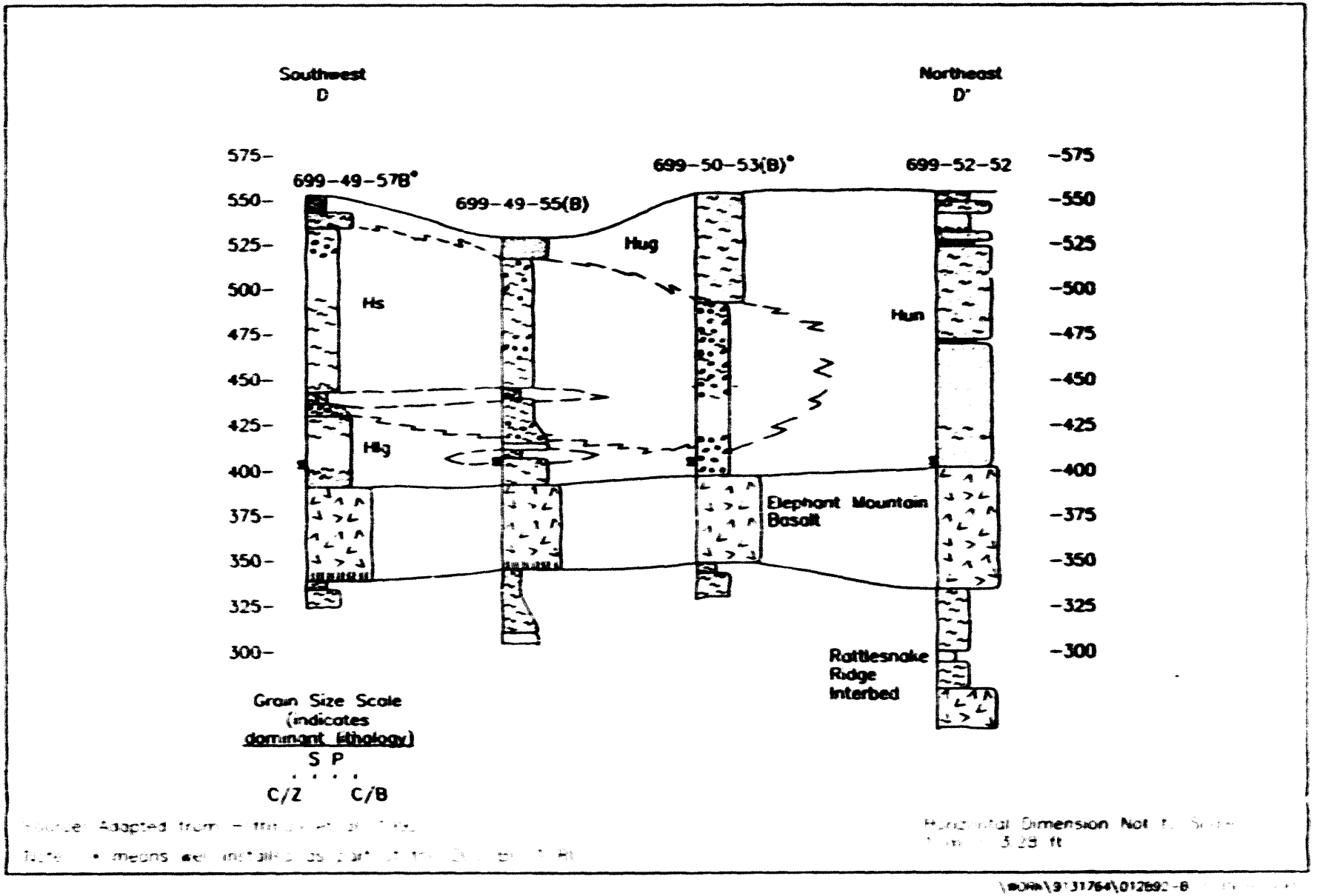

Figure 3-24. Cross Section D-D'

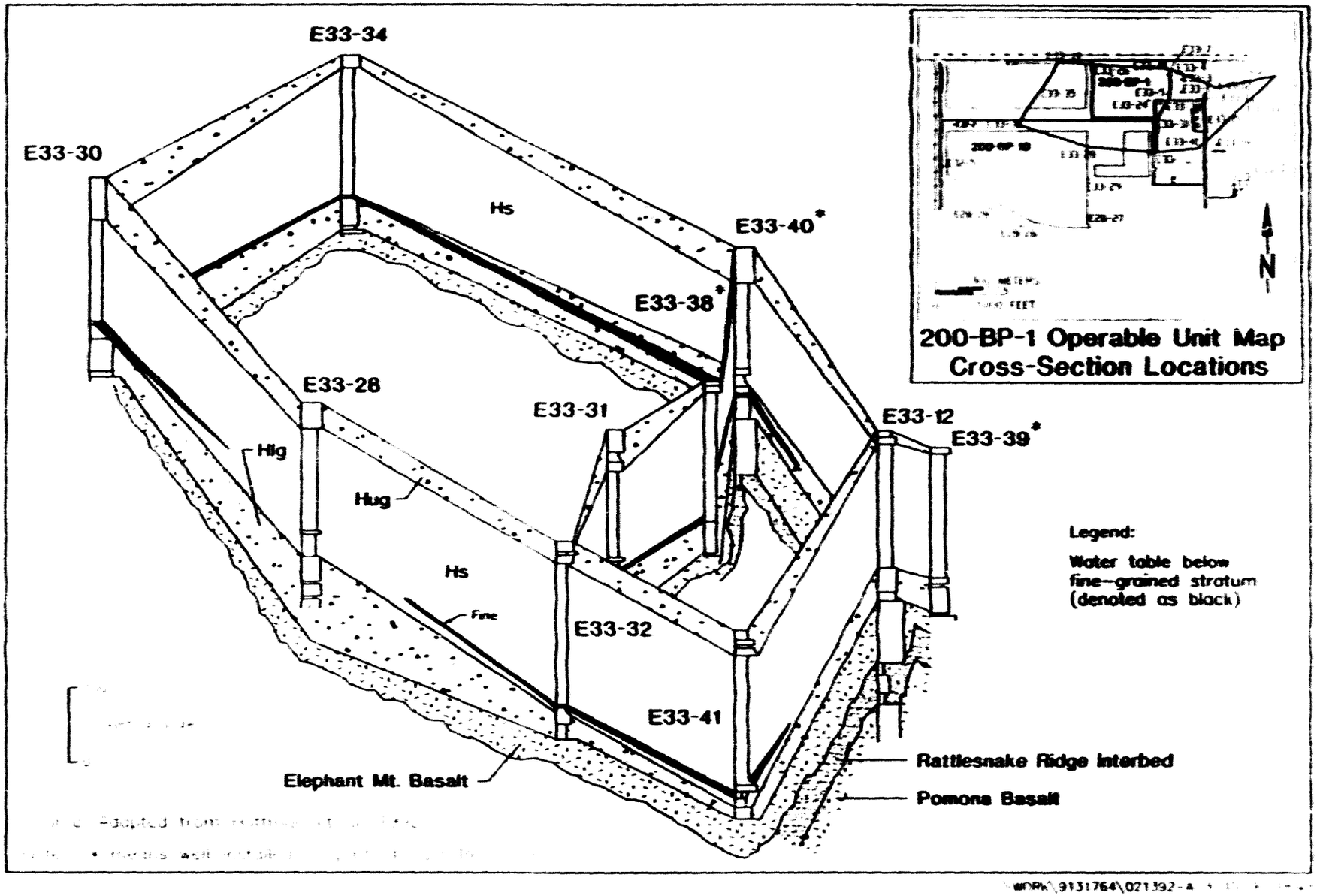

Figure 3-25. Geologic Fence Diagram for the 200-BP-1 Operable Unit.

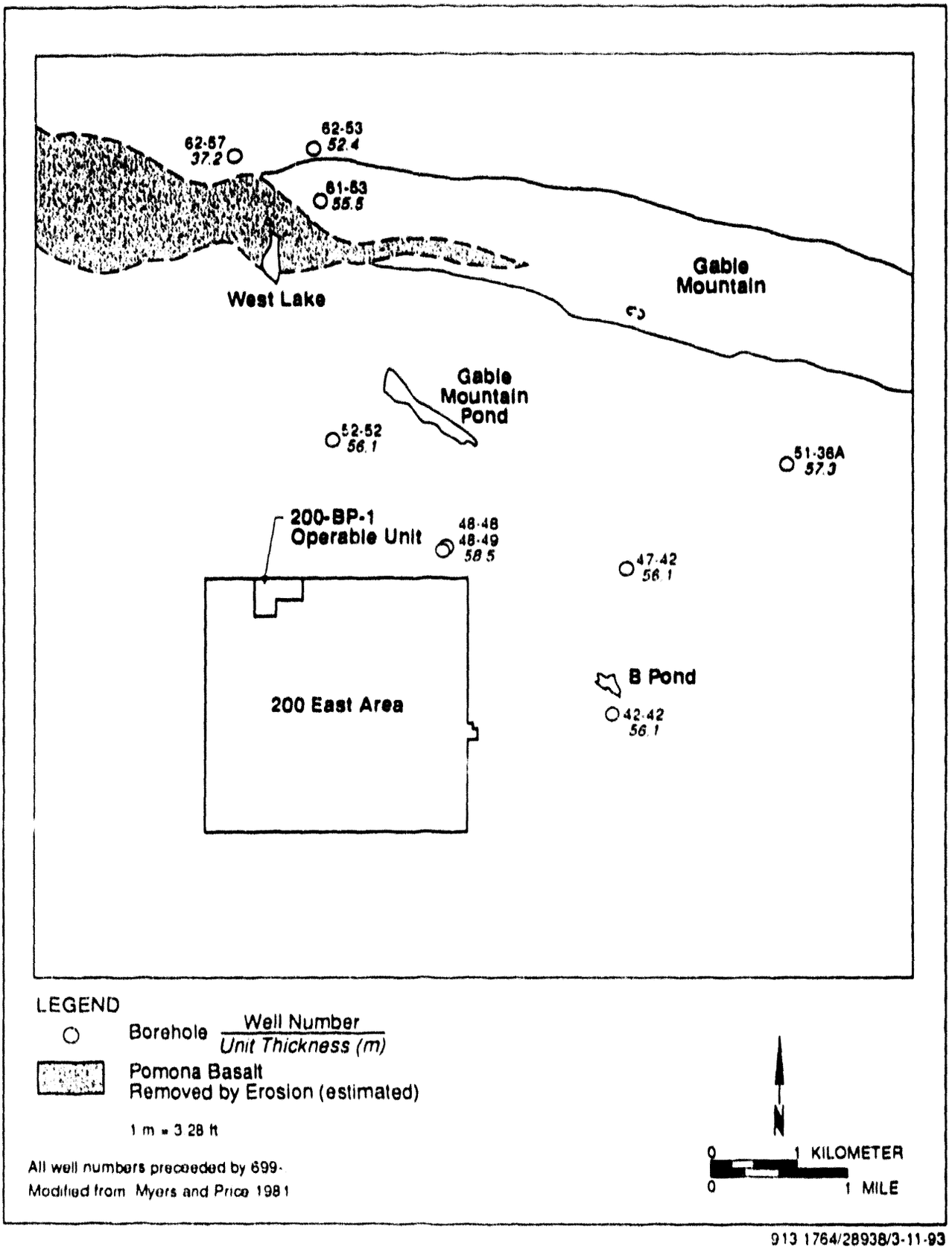

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Figure 3-26. Thickness Map of the Pomona Basalt.

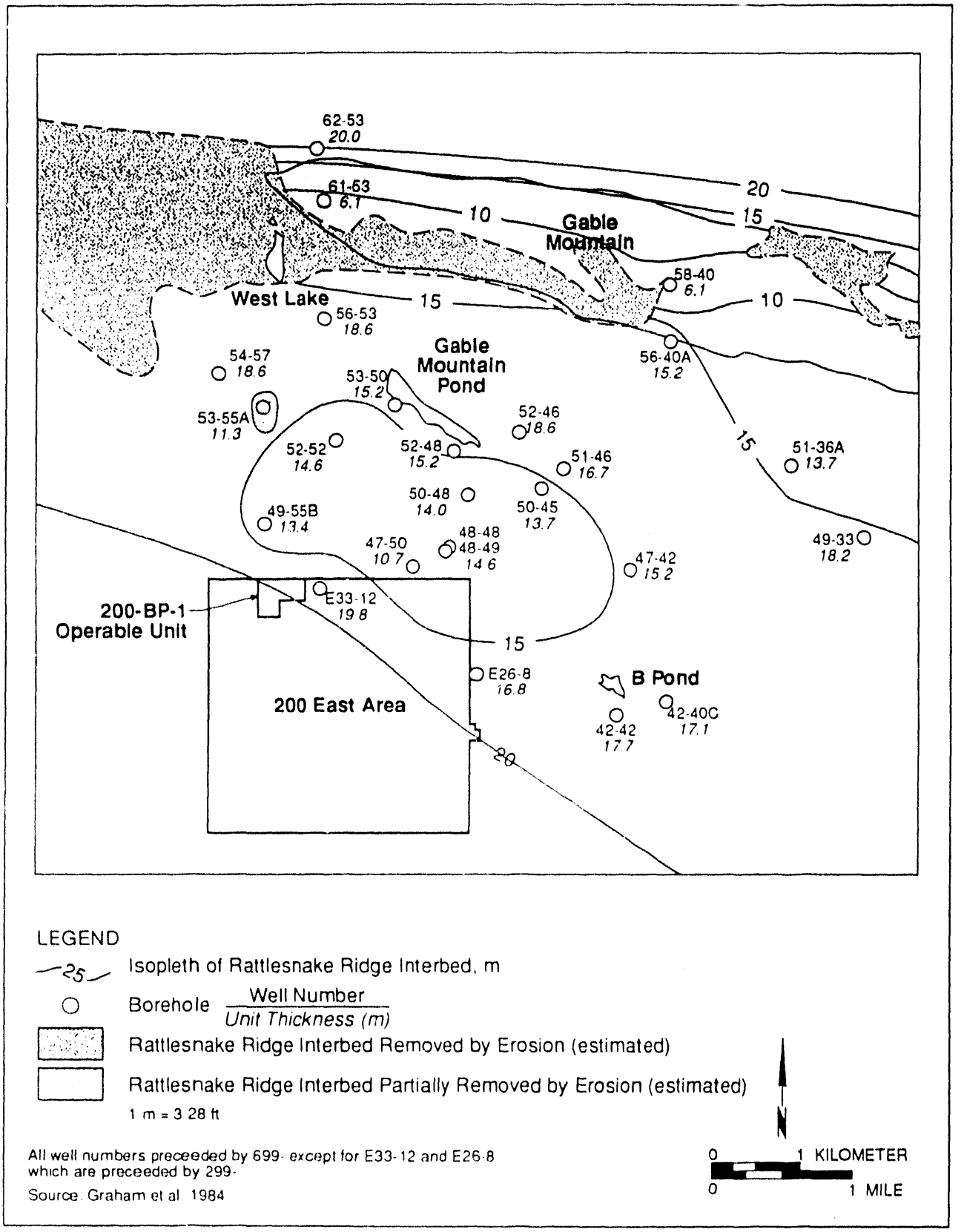

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Figure 3-27. Isopach Map of the Rattlesnake Ridge Interbed.

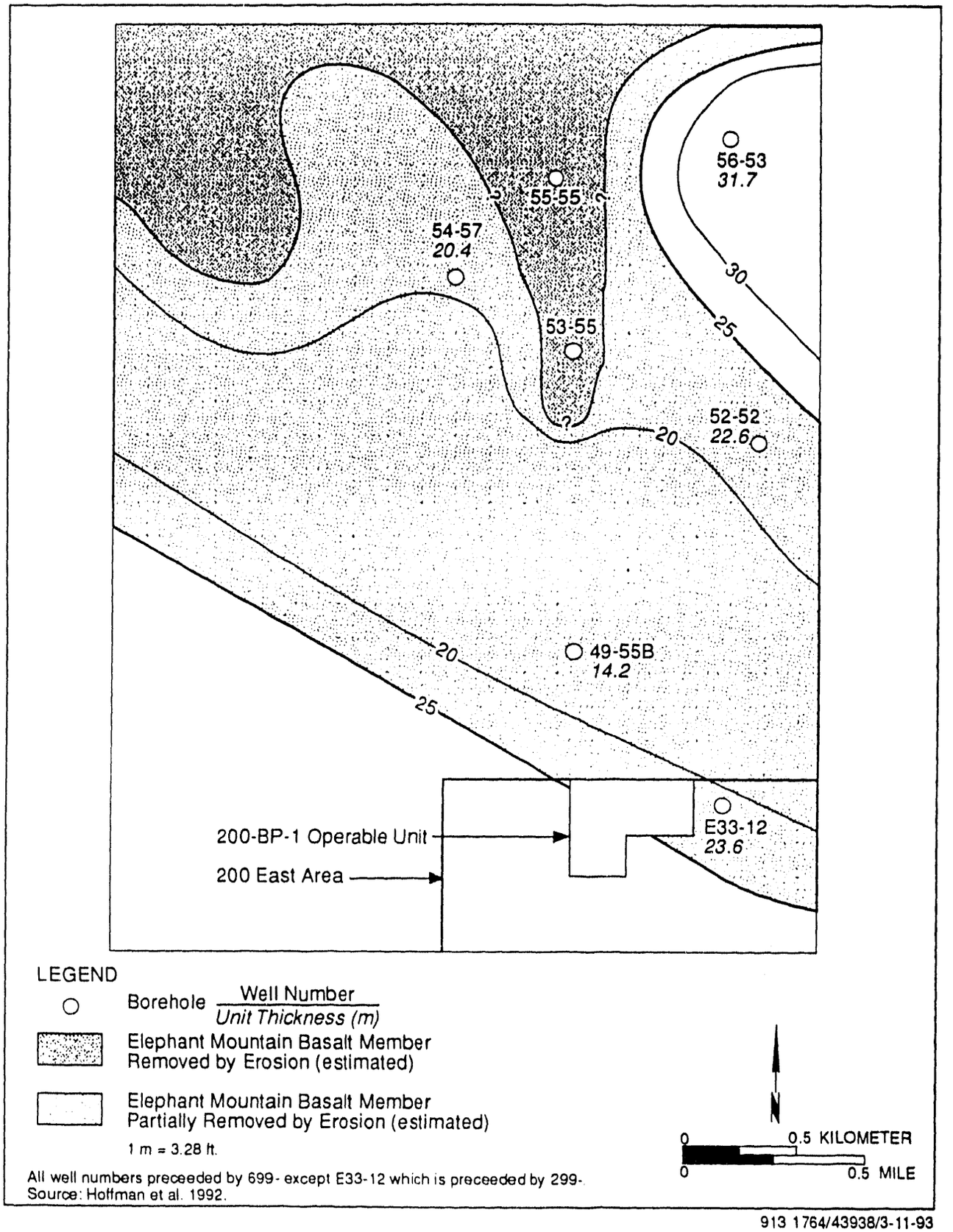
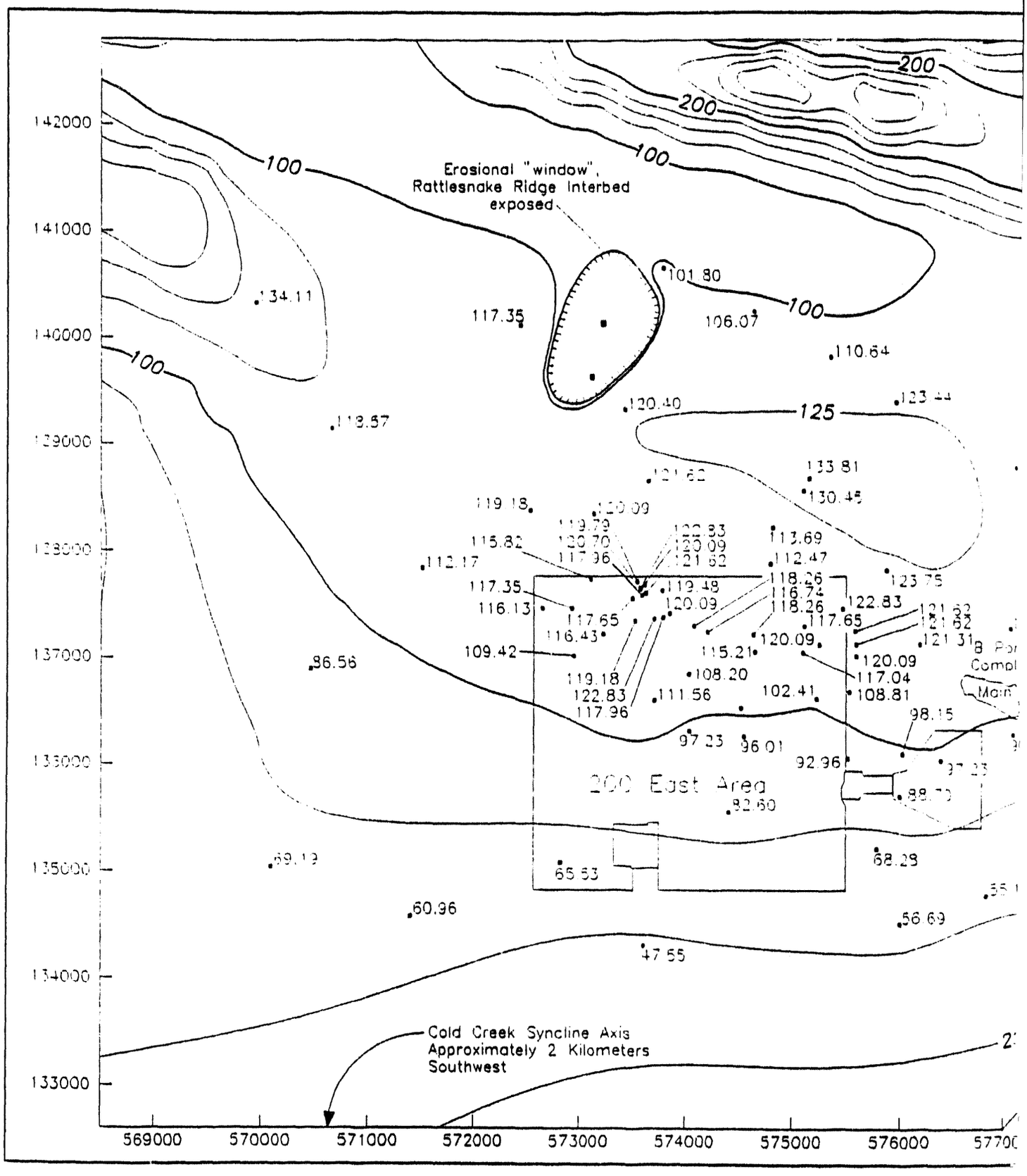
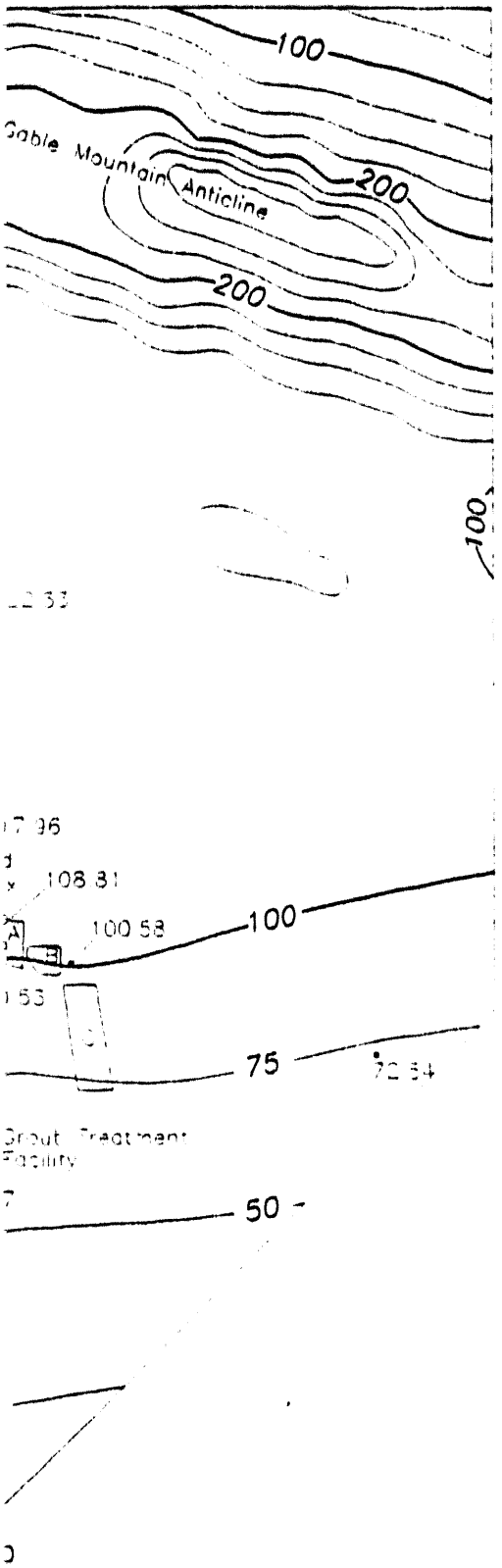


Figure 3-28. Isopach Map of the Elephant Mountain Basalt Member.


200
200
100
100
100
100
125
Erosional "window",
Rattlesnake Ridge Interbed
exposed
200 East Area
Cold Creek Syncline Axis
Approximately 2 Kilometers
Southwest
142000
141000
140000
139000
138000
137000
136000
135000
134000
133000
569000
570000
571000
572000
573000
574000
575000
576000
101.80
134.11
117.35
106.07
110.64
120.40
123.44
118.57
121.62
133.81
130.45
119.18
120.09
119.79
122.83
115.82
120.70
120.09
113.69
112.17
117.96
121.62
112.47
118.26
123.75
117.35
119.48
116.74
116.13
120.09
118.26
122.83
117.65
117.65
116.43
121.62
121.31
109.42
115.21
120.09
120.09
117.04
119.18
108.20
122.83
111.56
102.41
108.81
117.96
98.15
97.23
96.01
92.96
82.60
88.70
86.56
89.19
68.28
65.83
60.96
56.69
47.55



Legend:

- • Borehole
- ■ Borehole where Rattlesnake Ridge interbed encountered before Basalt
- 123.44 Elevation, in meters, of the Upper Surface of Basalt
- -200- Contour on Upper Surface of Basalt
- Depression

Contour Interval = 25 meters
1m = 3.28ft

Note: Grid in Lambert coordinates (meters), NAD83.

N

0 1000 2000 METERS
0 3000 6000 FEET

Source: Connelly et al. 1992.

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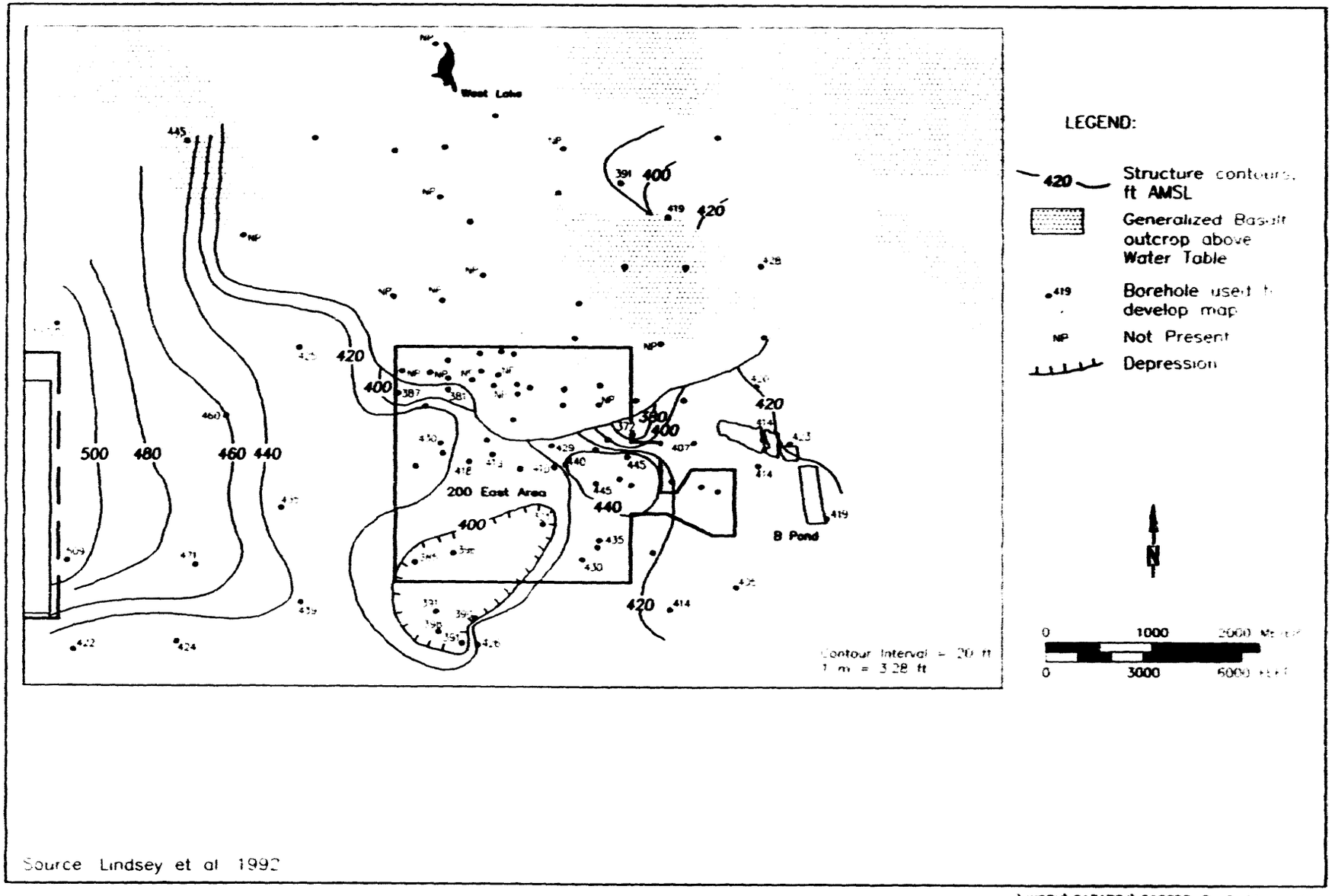
Figure 3-29. Structure Contour Map of the Top of Basalt Surface.



\WORK\9131764\012292-B

Figure 3-30. Structure Contour Map of the Top of the Ringold Formation.

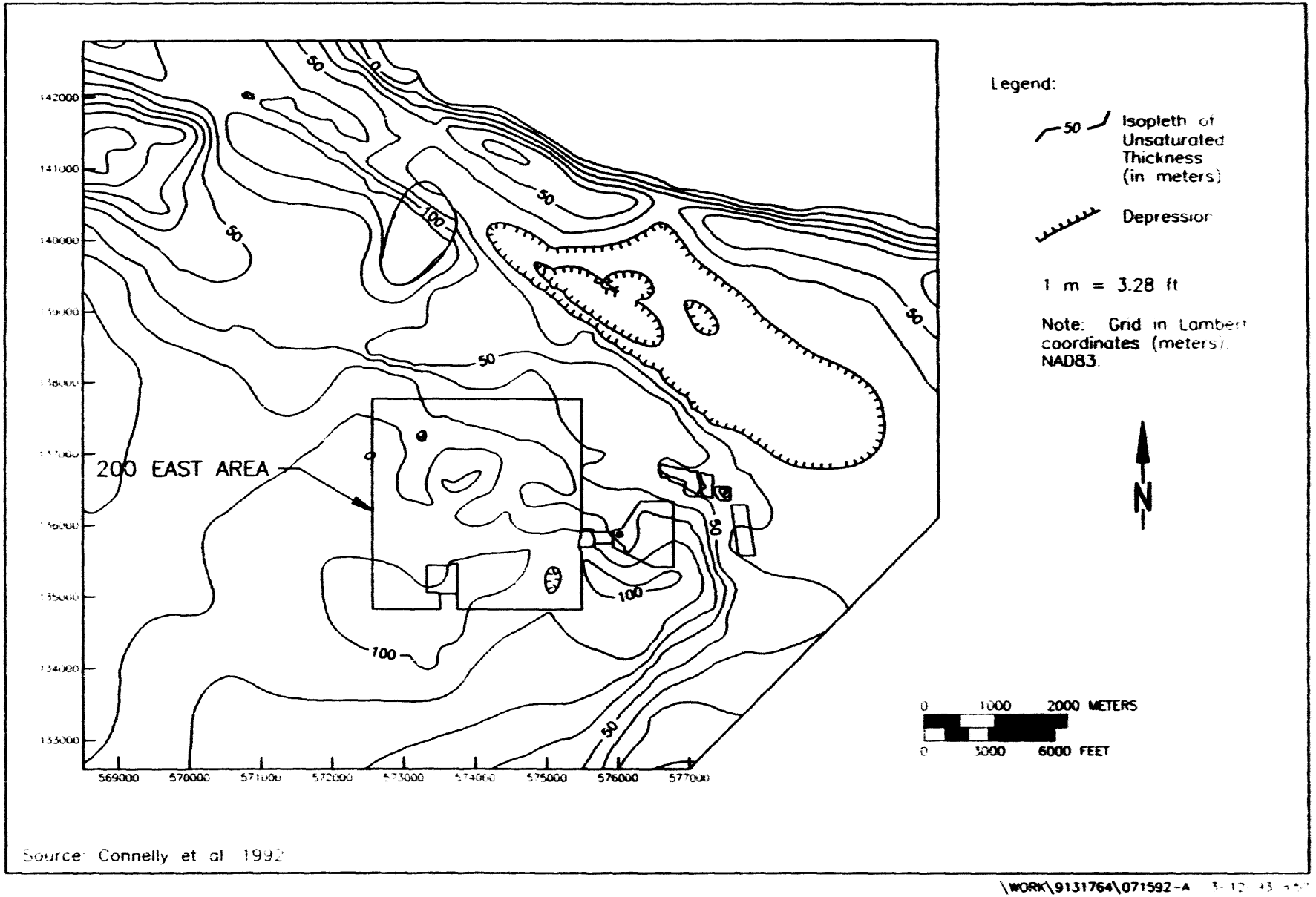
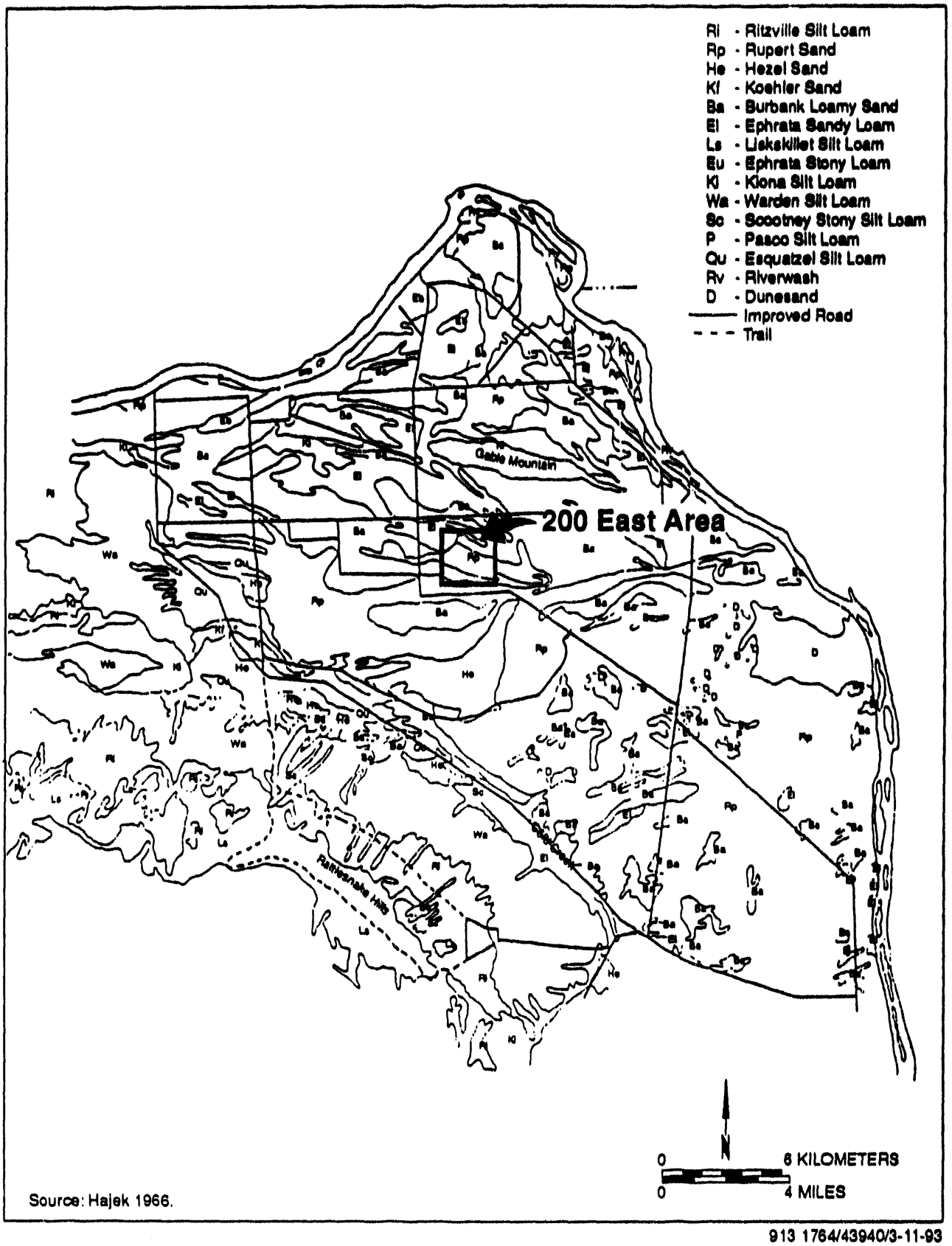
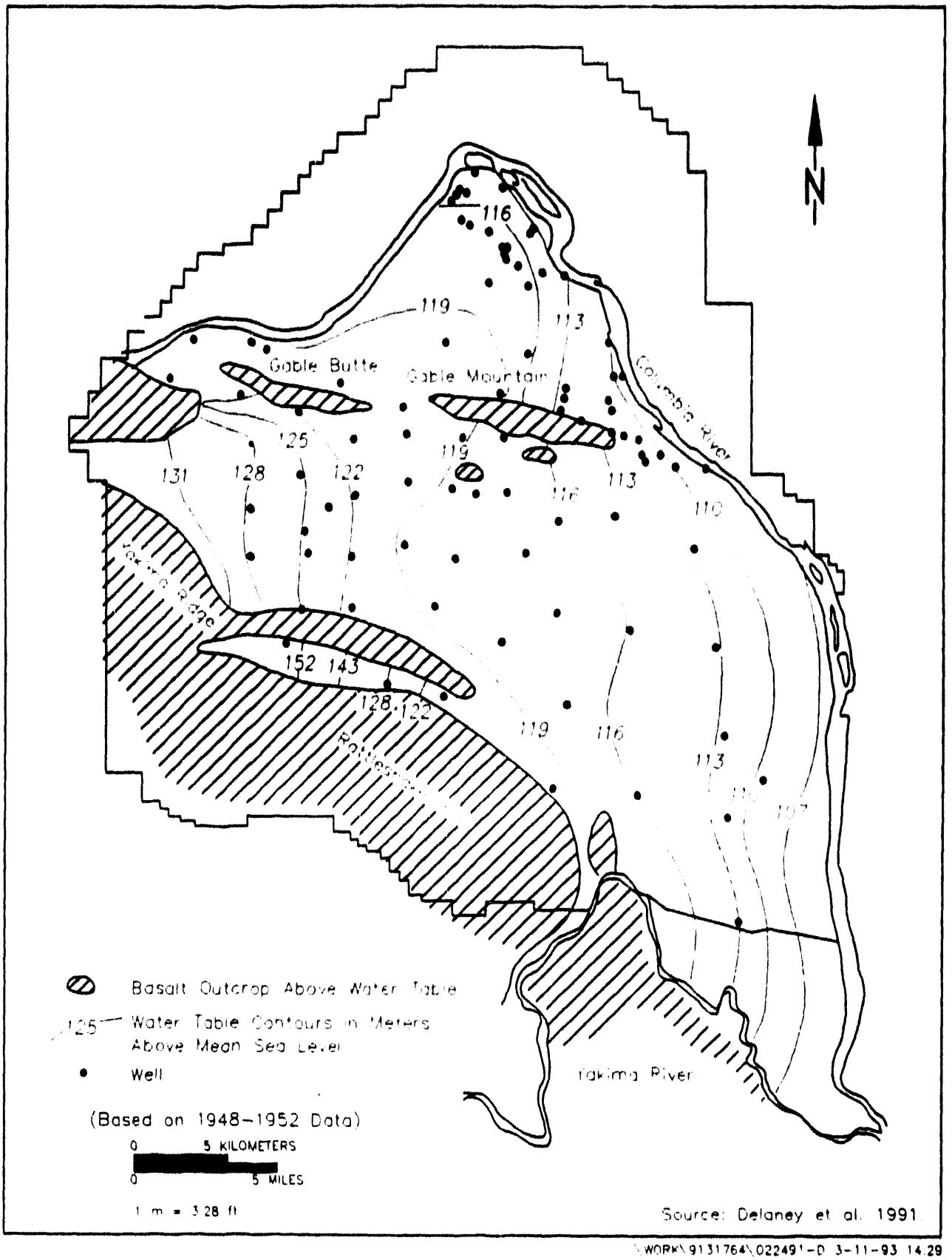


Figure 3-31. Isopach Map of the Total Thickness of the Hanford Formation.



913 1764/43940/3-11-93

Figure 3-32. Soil Map of the Hanford Project in Benton County, Washington.



\WORK\9131764\022491-D 3-11-93 14:29

Figure 3-33. Hanford Site Water Table Map, January 1944.

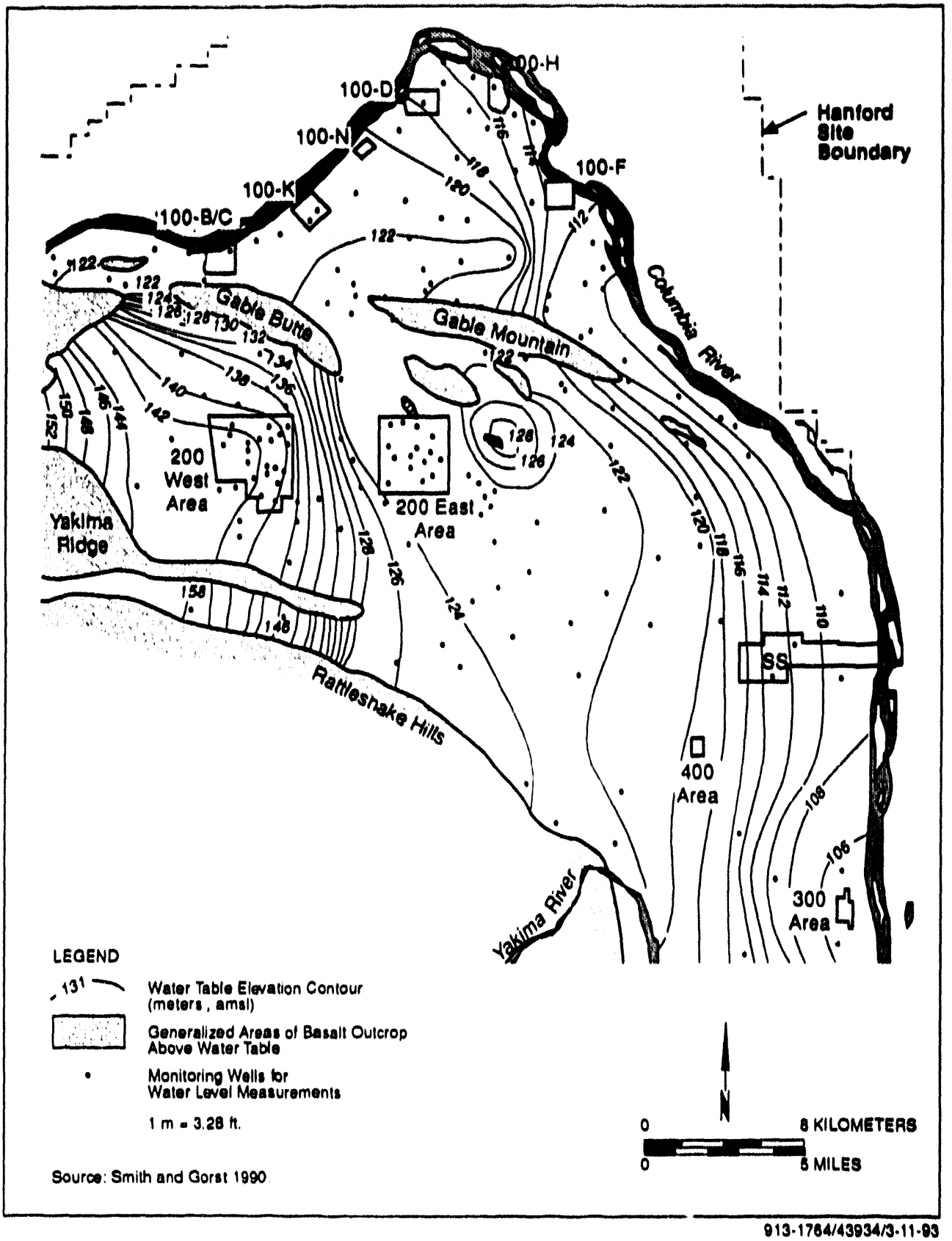


Figure 3-34. June 1989 Water Table Elevations for the Hanford Site.

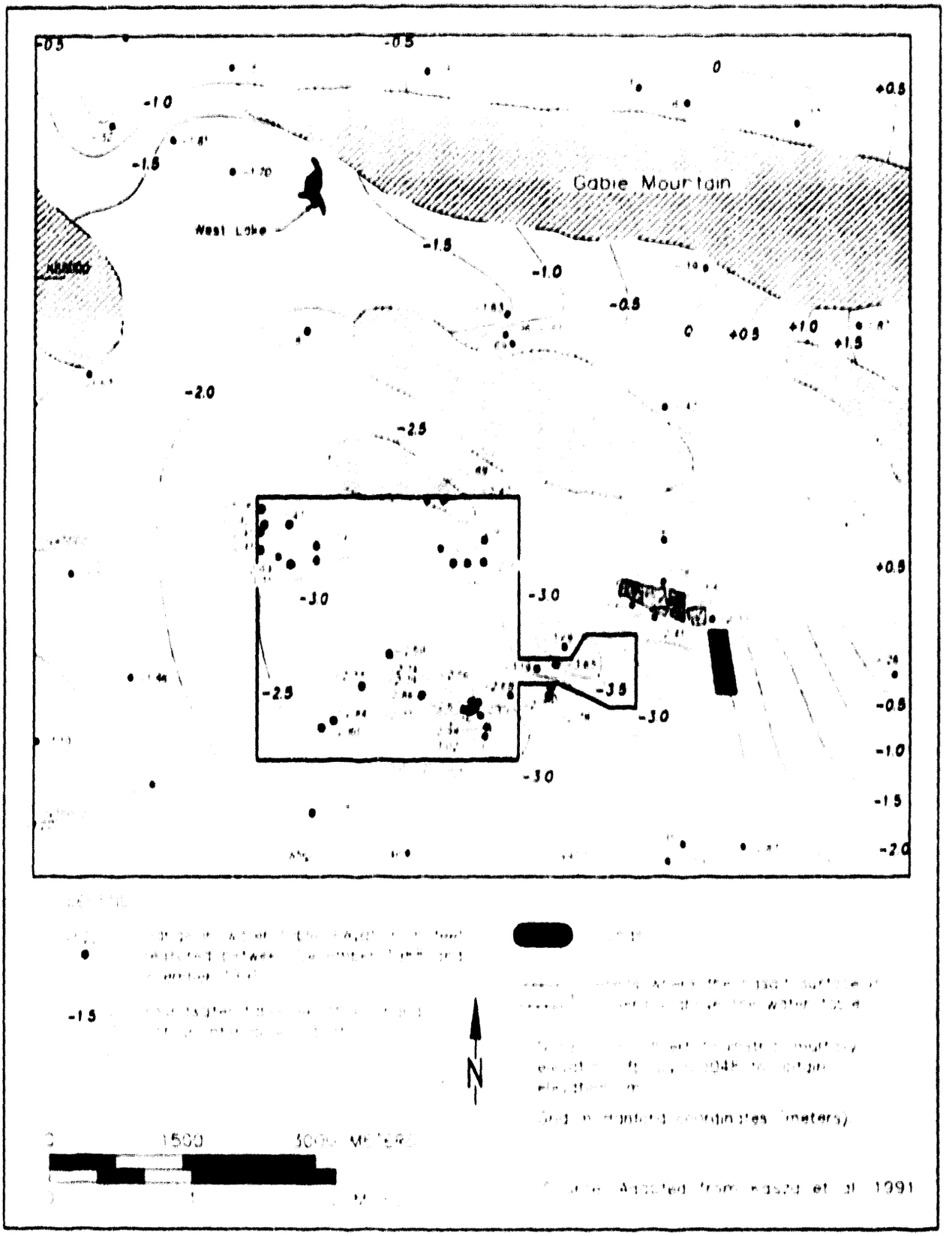


Figure 3-35. 200 East Areas Change in Water Table Elevation December 1988 to December 1991

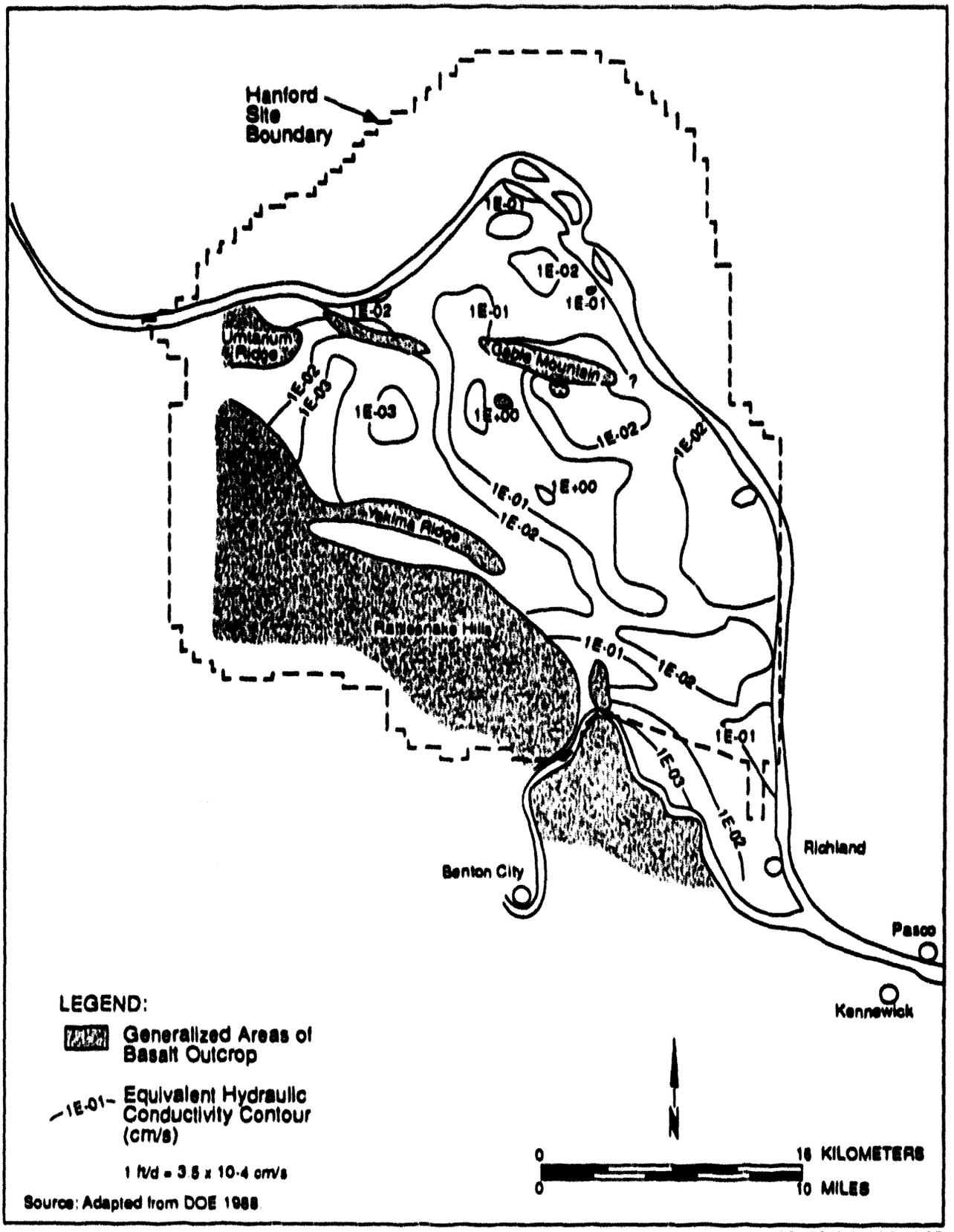


Figure 3-36. Areal Distribution of Hydraulic Conductivity for the Unconfined Aquifer.

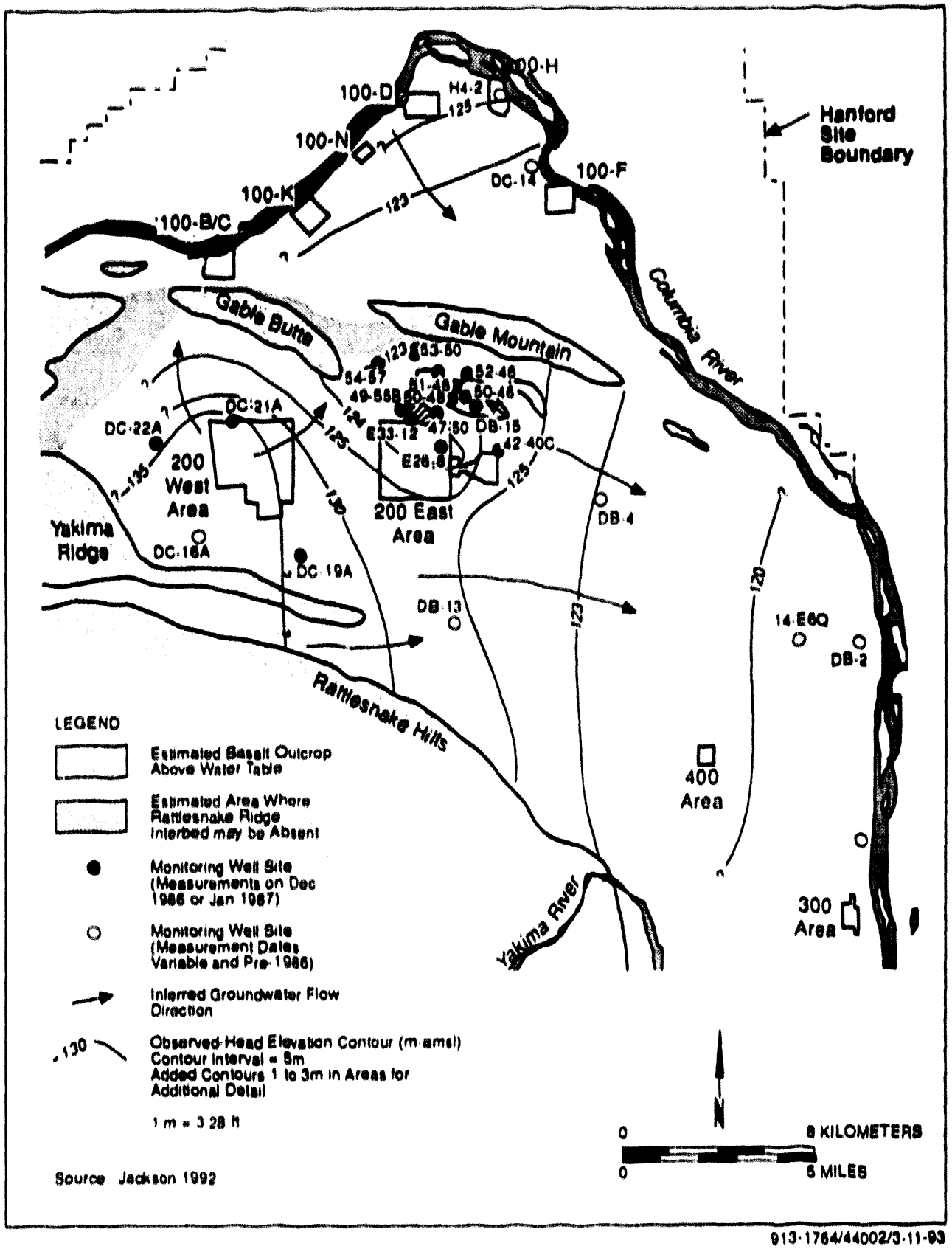


Figure 3-37. Potentiometric Map of the Rattlesnake Ridge Interbed, Hanford Site.

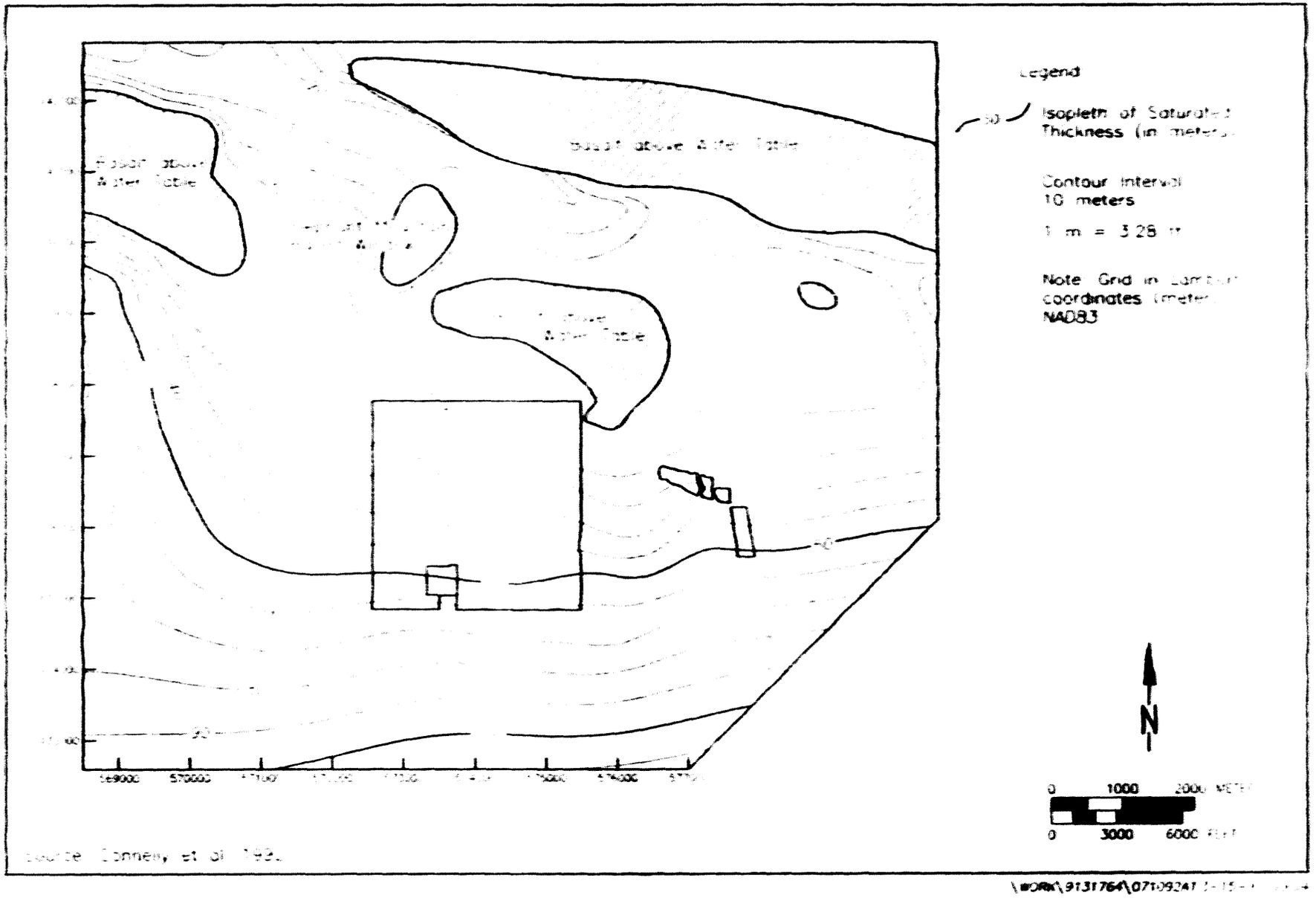
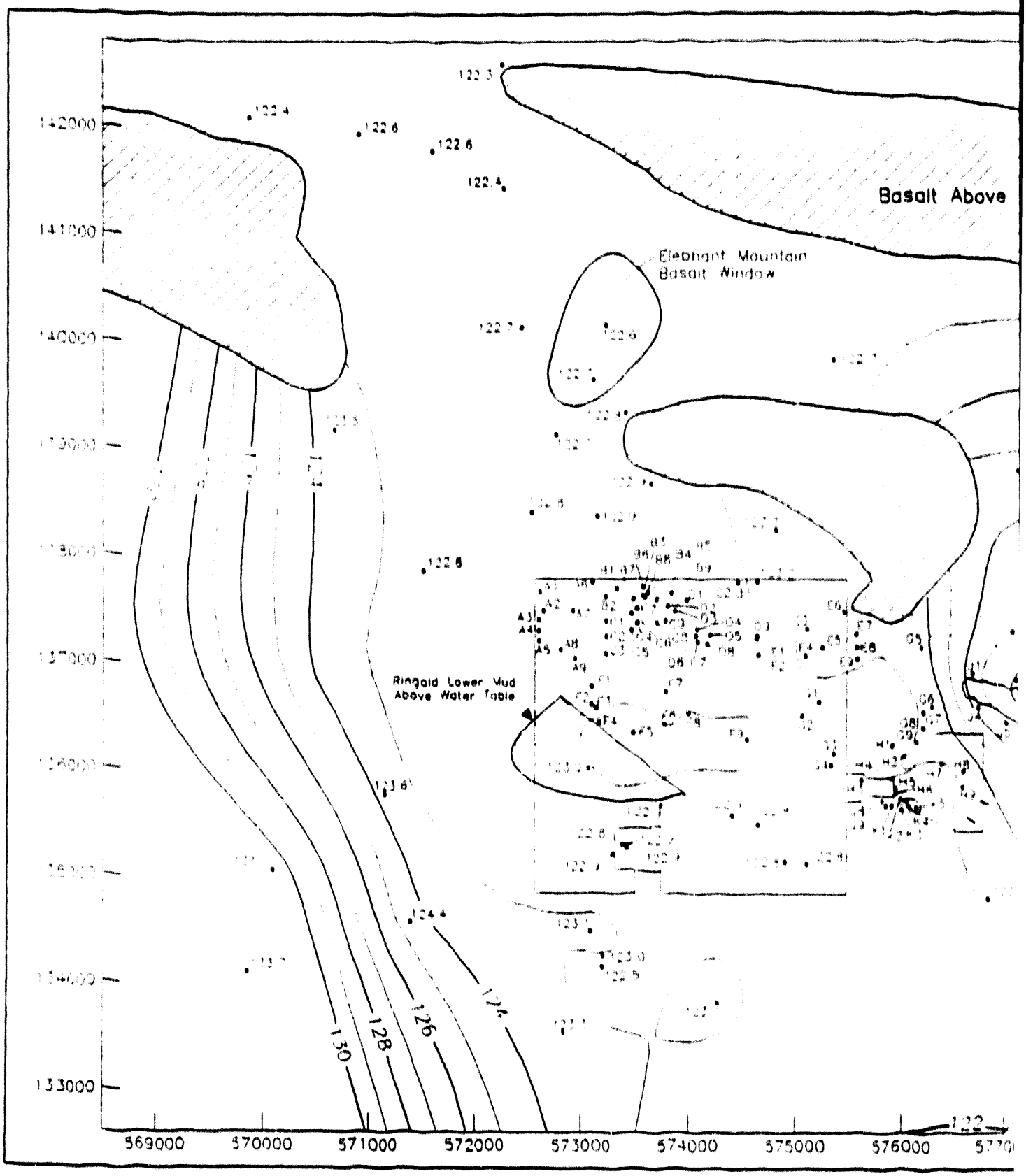
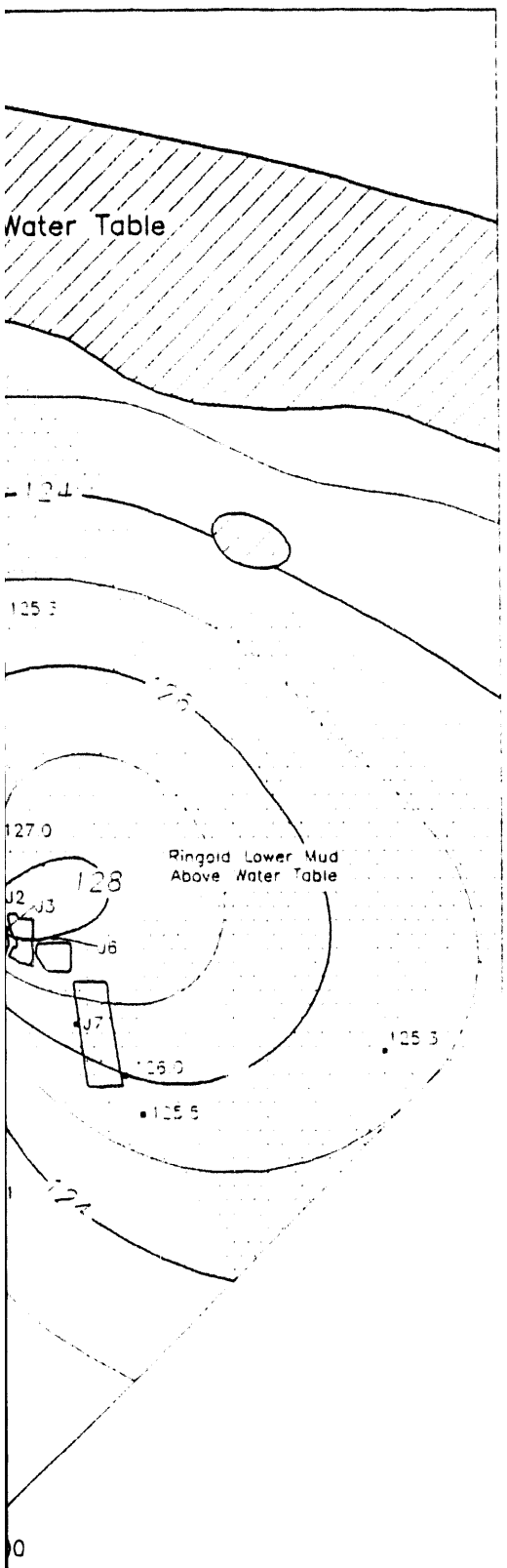


Figure 3-38. Isopach Map of the Uppermost Aquifer System.


Basalt Above
Elephant Mountain
Basalt Window
Ringold Lower Mud
Above Water Table
142000
141000
140000
139000
138000
137000
136000
135000
134000
133000
569000
570000
571000
572000
573000
574000
575000
576000
130
128
126
124



Legend:

124 Water Table Elevation (in meters)

Contour Interval = 1 meter

1 m = 3.28 ft

A1-122.9	B1-122.4	C1-122.8	D1-122.9	E1-122.9
A2-122.8	B2-122.8	C2-122.8	D2-123.3	E2-122.9
A3-122.8	B3-122.7	C3-122.8	D3-122.9	E3-123.0
A4-122.8	B4-122.8	C4-122.8	D4-122.8	E4-123.0
A5-122.8	B5-122.8	C5-122.9	D5-122.8	E5-123.1
A6-122.8	B6-122.8	C6-122.8	D6-122.4	E6-123.1
A7-122.8	B7-122.8	C7-122.8	D7-122.8	E7-123.1
A8-122.8	B8-122.8	C8-122.9	D8-122.9	E8-123.1
A9-122.9	B9-122.8	C9-122.9	D9-122.9	E9-123.1
F1-123.0	G1-122.9	H1-122.3	J1-126.6	K1-122.9
F2-123.0	G2-122.9	H2-123.0	J2-128.1	K2-122.8
F3-122.8	G3-122.8	H3-122.9	J3-128.1	K3-122.9
F4-123.1	G4-122.8	H4-123.0	J4-126.4	K4-123.0
F5-123.2	G5-122.7	H5-122.9	J5-126.8	K5-123.0
F6-123.0	G6-123.1	H6-123.0	J6-128.0	
F7-122.7	G7-123.1	H7-123.0	J7-126.5	
F8-123.0	G8-123.1	H8-123.1	J8-123.0	
F9-123.3	G9-123.1	H9-123.2	J9-123.0	

Note: Grid in Lambert coordinates (meters), NAD83.

N

0 1000 2000 METERS

0 3000 6000 FEET

Source: Connelly et al. 1992.

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Figure 3-39. December 1991 Water Table Elevations.

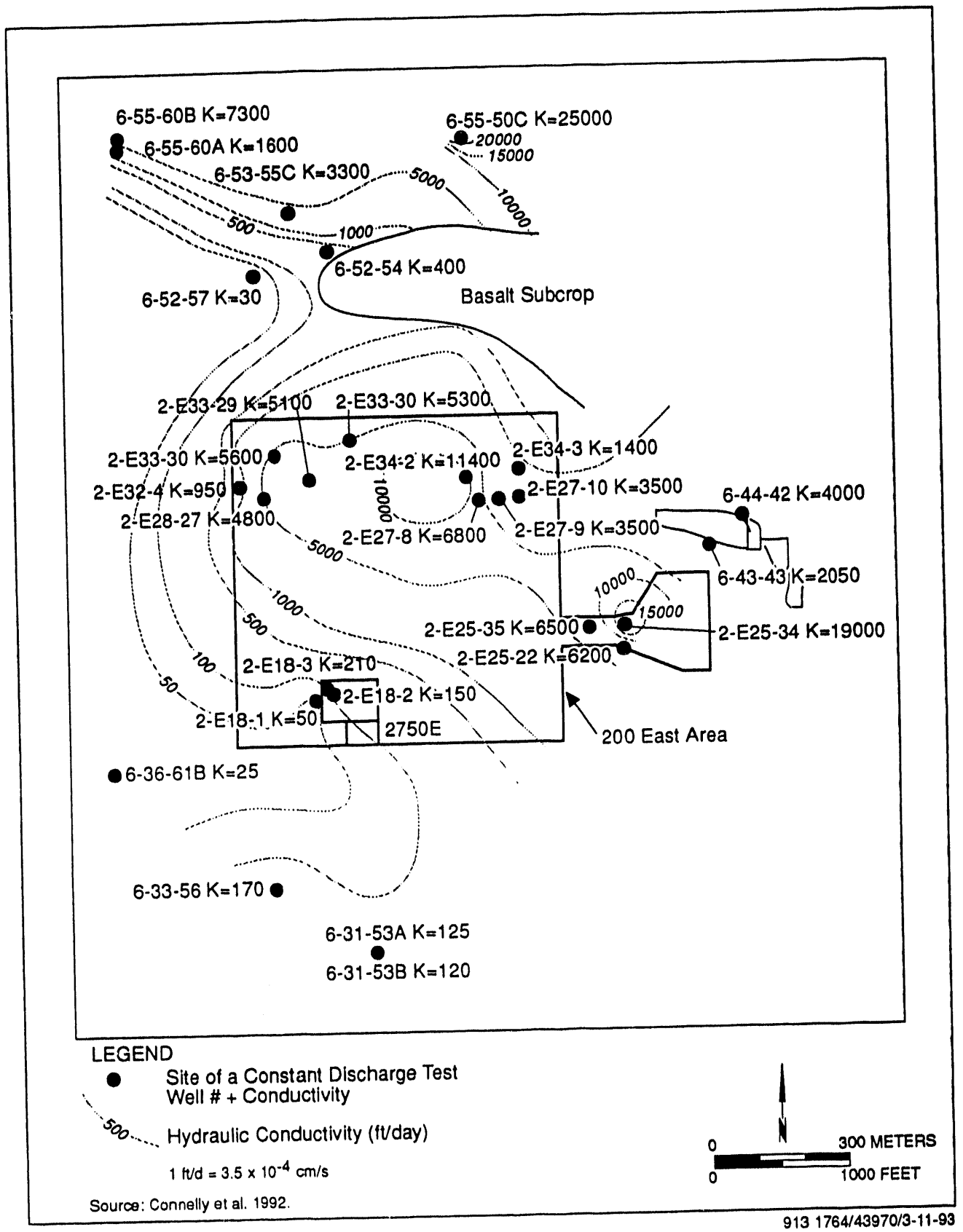

Figure 3-40. Hydraulic Conductivity Contours for the Uppermost Aquifer System.

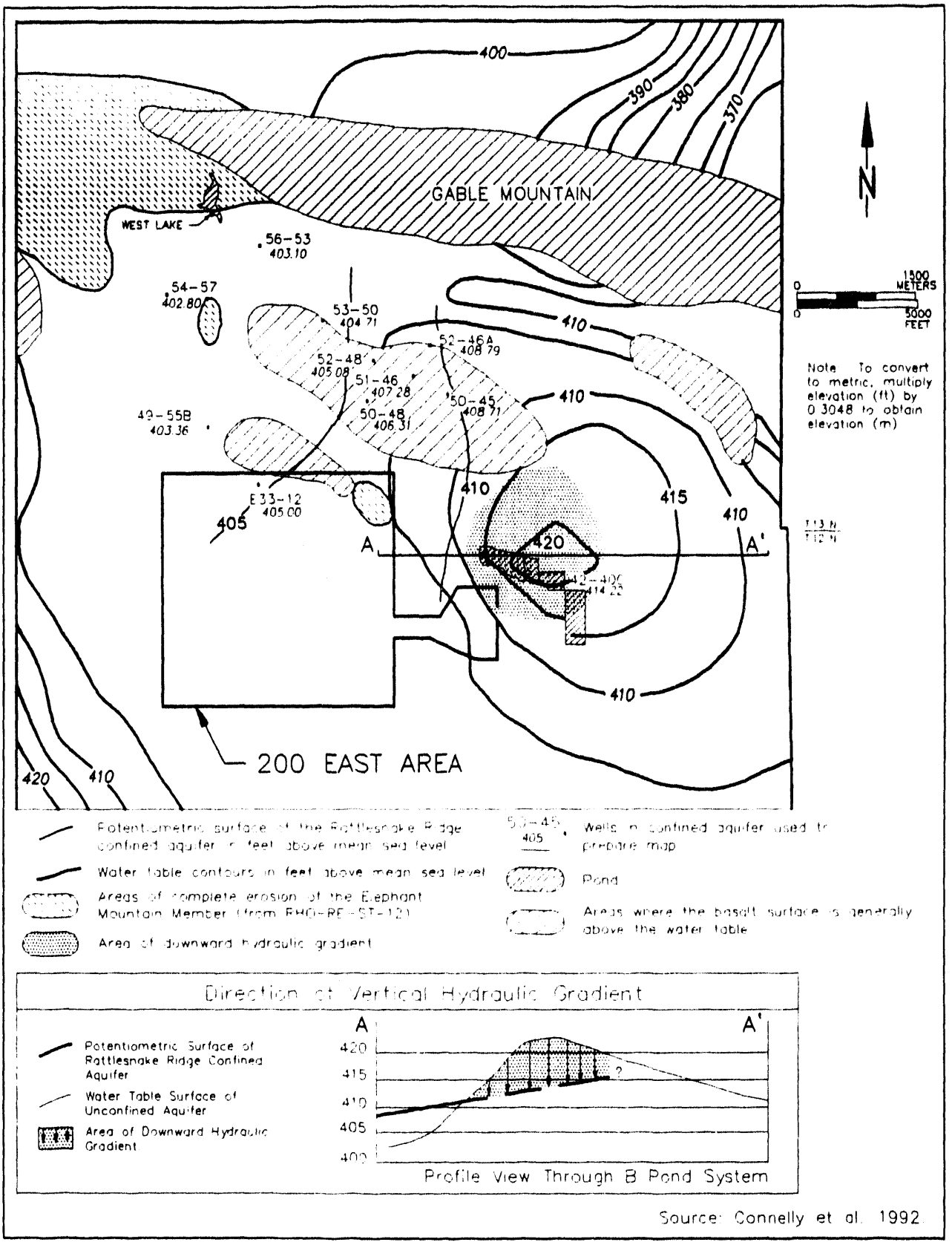

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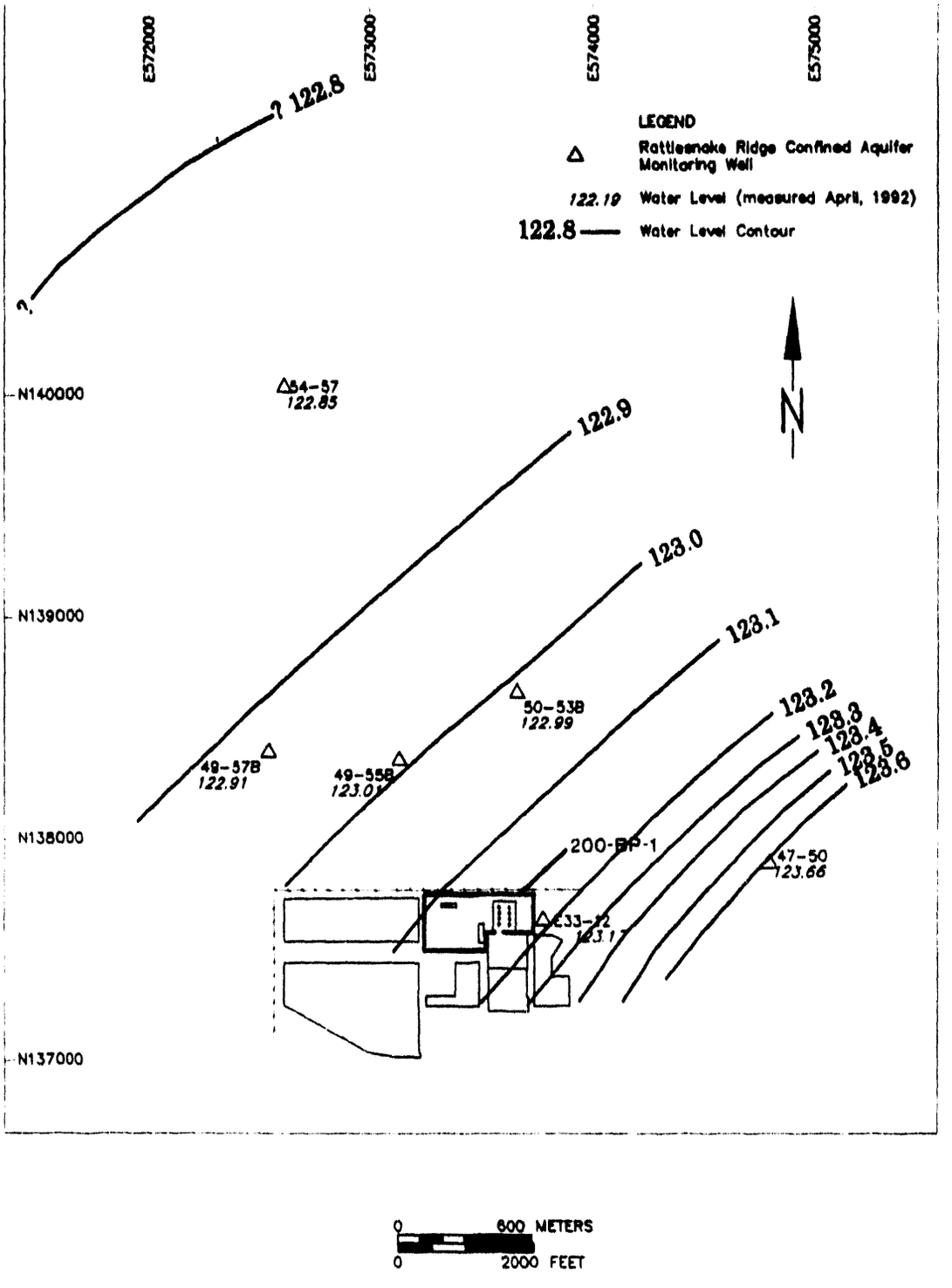
Figure 3-41. Comparison of Potentiometric Surface of the Rattlesnake Ridge Confined Aquifer with the Water Table of the Unconfined Aquifer, June 1991.



Note: Grid in Lambert coordinates (m), NAD83.

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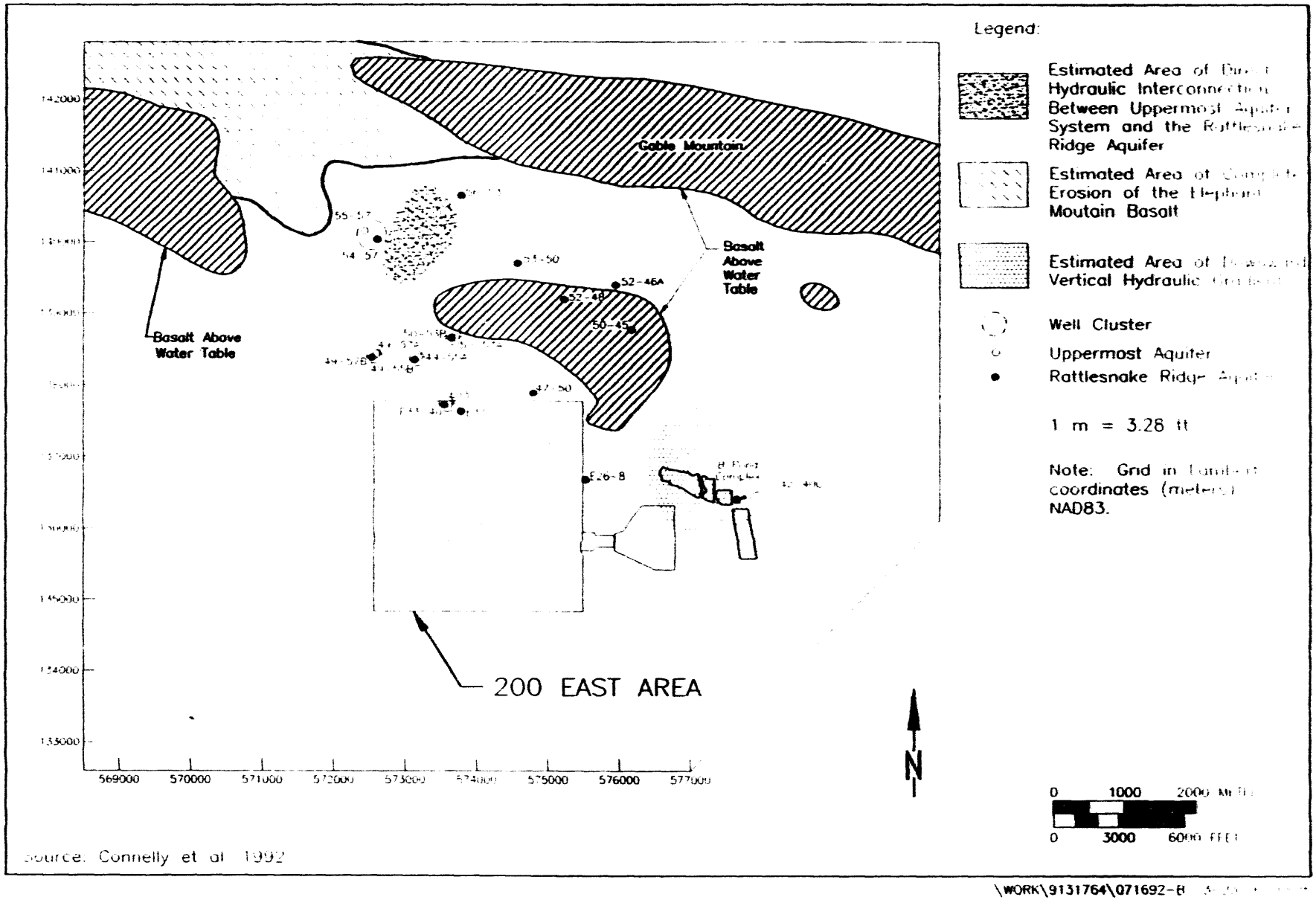
Figure 3-42. Potentiometric Surface of the Rattlesnake Ridge Confined Aquifer in the Study Area.



\WORK\9131764\071692-B

Figure 3-43. Rattlesnake Ridge Aquifer Monitoring Wells in the 200-BP-1 Study Area.

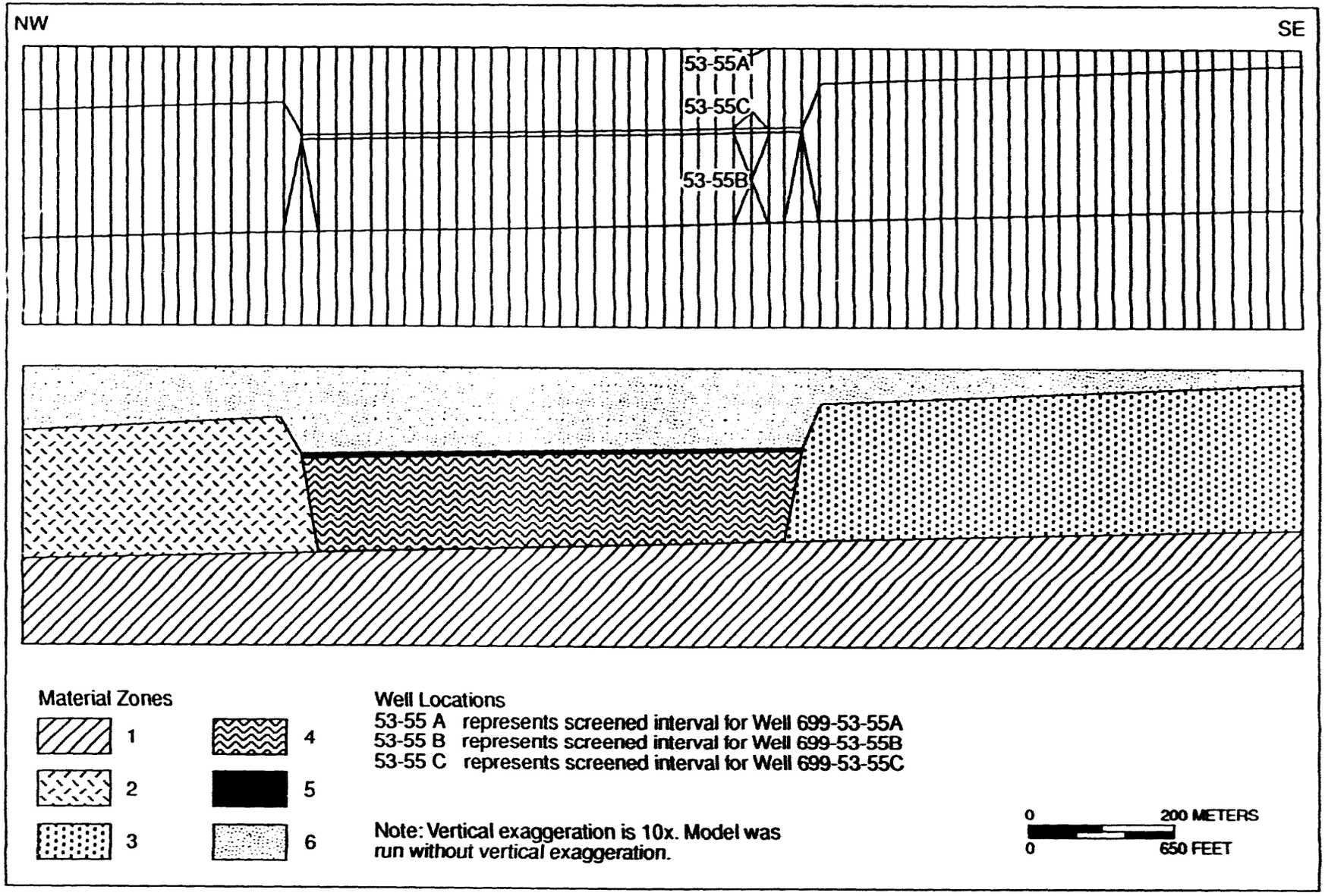


Figure 3-44. Finite-Element Mesh and Material Zones for the Erosional Window Model.

Simulation		Material Zone						h_C-h_B (ft)
		1	2	3	4	5	6	
A	K_X (ft/d)	8	10^{-7}	10^{-7}	1000	1000	1000	<0.009
	K_Z (ft/d)	0.8	10^{-7}	10^{-7}	200	200	200	
B	K_X (ft/d)	8	10^{-7}	10^{-7}	1000	1000	1000	<0.009
	K_Z (ft/d)	0.8*	10^{-7}	10^{-7}	200	200	200	
C	K_X (ft/d)	8	10^{-7}	10^{-7}	10000*	1000	1000	<0.009
	K_Z (ft/d)	0.8	10^{-7}	10^{-7}	2000*	200	200	
D	K_X (ft/d)	8	10^{-7}	10^{-7}	1000	0.1*	1000	0.019
	K_Z (ft/d)	0.8	10^{-7}	10^{-7}	200	0.01*	200	
E	K_X (ft/d)	8	1000*	10^{-7}	1000	0.1*	1000	0.067
	K_Z (ft/d)	0.8	200*	10^{-7}	200	0.01*	200	

h_B = hydraulic head at well 699-53-55B
h_C = hydraulic head at well 699-53-55C
K_X = horizontal hydraulic conductivity (ft/d)
K_Z = vertical hydraulic conductivity (ft/d)
*Differs from the hydraulic conductivity used in simulation A.
Actual h_C - h_B is approximately 0.25 ft. (Connelly et al. 1992)

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Figure 3-45. Simulation Results for the Erosional Window Finite-Element Model.

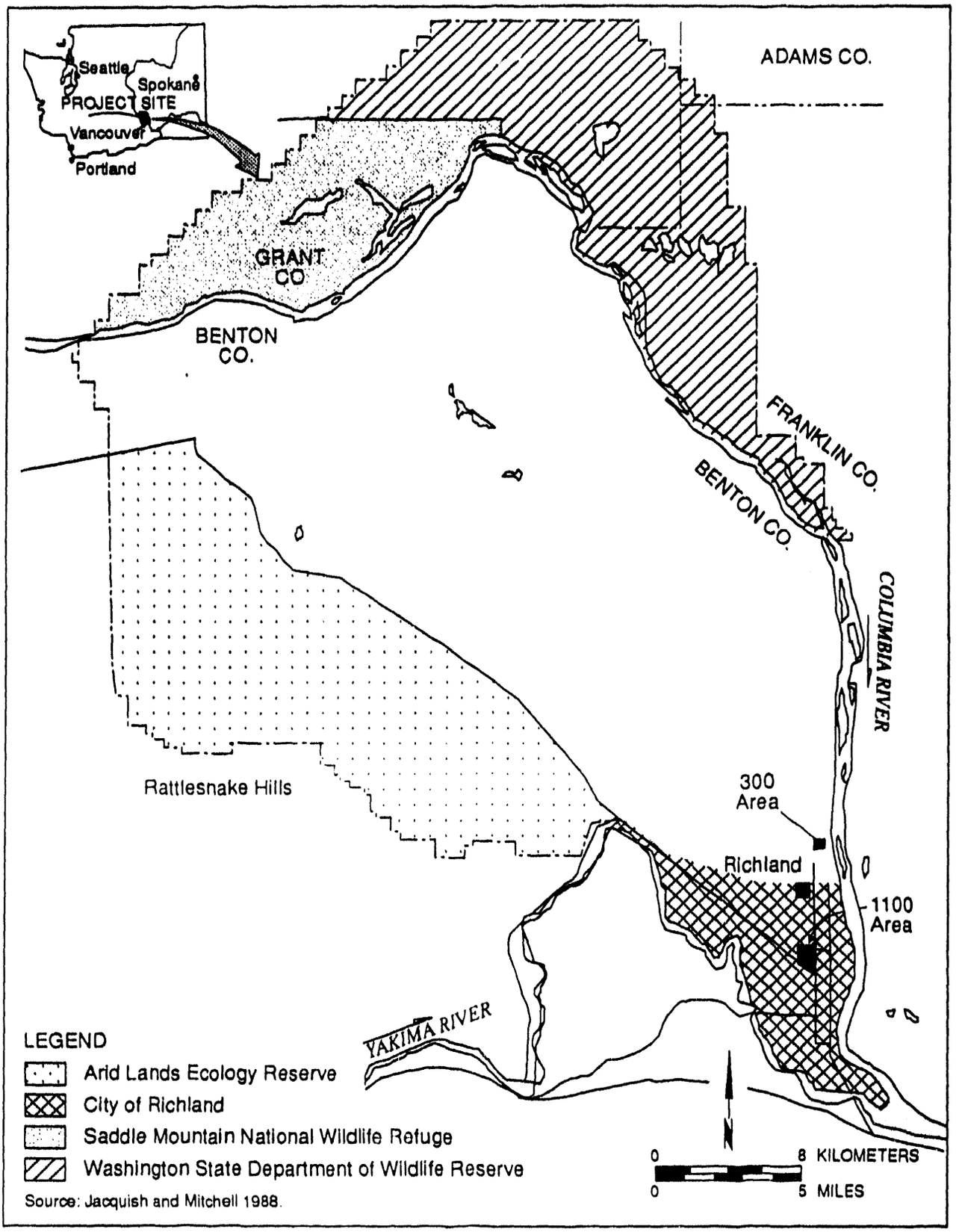


Figure 3-46. Locations of Wildlife and Ecological Land Reserves.

Table 3-1. Sites Included in the 200-BP-1 Operable Unit.

Cribs	Period of Use	Waste Source	Waste Volume/Area
216-B-43	11/54 to 11/54	TBP[a] supernatant from 221-U building	2.12×10^6 L
216-B-44	11/54 to 3/55	TBP supernatant from 221-U building	5.6×10^6 L
261-B-45	4/55 to 6/55	TBP supernatant from 221-U building	4.92×10^6 L
216-B-46	9/55 to 12/55	TBP supernatant from 221-U building	6.7×10^6 L
216-B-47	9/55 to 9/55	TBP supernatant from 221-U building	3.7×10^6 L
216-B-48	11/55 to 7/57	TBP supernatant from 221-U building	4.09×10^6 L
216-B-49	11/55 to 12/55	TBP supernatant from 221-U building	6.7×10^6 L
216-B-50	3/65 to 1/74	ITS[b] 1 tank condensate	54.8×10^6 L
216-B-57	2/68 to 6/73	ITS 2 tank condensate	84.4×10^6 L
216-B-61	Not applicable	No documented discharge of waste	0 L
Unplanned Release		**Date**	
UN-200-E-9		September 15, 1955	41,635 L
UN-200-E-110		August 7, 1955	2,320 m^2
UN-200-E-63		Unknown	Unknown
UN-200-E-89		Contaminant spreading occurred gradually over a number of years, pre-1984 to 1991. Initial date not known.	4.3 ha

[a]Tributyl phosphate
[b]In-tank solidification unit
Source: DOE-RL (1992a)

Table 3-2. Average Return Period (R) and Existing Record (ER) for Various Precipitation Amounts and Intensity During Specified Time Periods at Hanford. (Based on Extreme Value Analysis of 1947 through 1969 Records)

	Amount (Inches)							Intensity (Inches Per Hour)						
	Time Period							Time Period						
R Years	20 Min	60 Min	2 Hrs	3 Hrs	6 Hrs	12 Hrs	24 Hrs	20 Min	60 Min	2 Hrs	3 Hrs	6 Hrs	12 Hrs	24 Hrs
2	0.16	0.26	0.30	0.36	0.48	0.62	0.72	0.49	0.26	0.15	0.12	0.08	0.052	0.030
5	0.24	0.40	0.48	0.55	0.77	0.95	1.06	0.72	0.40	0.24	0.18	0.13	0.079	0.044
10	0.37	0.50	0.59	0.67	0.96	1.17	1.28	1.1	0.50	0.30	0.22	0.16	0.098	0.053
25	0.47	0.62	0.74	0.83	1.21	1.45	1.56	1.4	0.62	0.37	0.28	0.20	0.121	0.065
50	0.53	0.72	0.85	0.96	1.40	1.66	1.77	1.6	0.72	0.42	0.32	0.23	0.138	0.074
100	0.60	0.81	0.96	1.07	1.59	1.87	1.99	1.8	0.81	0.48	0.36	0.27	0.156	0.083
250	0.68	0.93	1.11	1.22	1.82	2.13	2.26	2.0	0.93	0.55	0.41	0.30	0.177	0.094
500	0.73	1.02	1.22	1.33	2.00	2.34	2.47	2.2	1.02	0.61	0.44	0.33	0.195	0.103
1000	0.80	1.11	1.33	1.45	2.20	2.55	2.68	2.4	1.11	0.67	0.48	0.37	0.212	0.112
ER	*	0.59	0.88	1.08	1.68	1.88	1.91	*	0.59	0.44	0.36	0.28	0.157	0.080
		6/12	10/1	10/1	10/1-2	10/1-2	10/1-2		6/12	10/1	10/1	10/1-2	10/1-2	10/1-2
Date	–	1969	1957	1957	1957	1957	1957	–	1969	1957	1957	1957	1957	1957

* No records have been kept for time periods of less than 60 minutes.

Source: Stone et al. (1983)

Table 3-3. Percent Frequency Distribution of Wind Direction at the 200 East Area Telemetry Site Record From Month 2 1979 to Month 12 1988.

Speed MPH	N	NNE	NE	ENE	E	ESE	SE	SSE	S	SSW	SW	NSW	W	WNW	NW	NNW	CALM	TOTAL
Calm																	0.9	0.9
1-3	1.9	1.8	2.5	1.9	2.0	1.8	1.6	1.2	1.2	0.8	0.9	1.0	1.6	1.9	2.1	1.9	0	26
4-7	1.8	1.4	1.0	1.0	1.7	2.0	2.6	2.2	1.5	1.0	1.4	2.1	4.4	6.8	4.8	2.3	0.0	37.9
8-12	0.7	0.6	0.2	0.1	0.2	0.3	0.6	1.0	0.7	0.4	0.9	1.7	4.0	7.6	2.6	0.6	0.0	22.4
13-18	0.2	0.2	0.1	0.0	0.0	0.0	0.0	0.2	0.2	0.3	0.8	1.2	1.2	3.3	1.4	0.1	0.0	9.1
19-24	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.4	0.5	0.2	1.0	0.6	0.0	0.0	3.0
25-31	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.1	0.1	0.0	0.0	0.6
32-38	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
39-46	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0
GT-46	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
TOTAL	4.6	4.0	3.9	3.1	3.9	4.2	4.8	4.6	3.6	2.6	4.4	6.7	11.4	20.7	11.7	4.9	0.9	100.0
Av. Speed	5.2	5.1	3.9	3.5	3.8	4.3	4.9	5.9	5.8	7.3	9.2	9.6	7.9	9.2	8.1	4.8		

Source: HMS Telemetry System Database
PNL, Atmospheric Sciences Department, Hanford Meteorologic Station

Data obtained through Personal Correspondence with Mr. Ora Gifford (HMS), included in Appendix G.

Table 3-4. Hanford Sequence Elevations and Thicknesses. (Sheet 1 of 2)

Borehole #	Hug T[1]	Hug TH[2]	Hs T[1]	Hs TH[2]	Hlg T[1]	Hlg TH[2]	Total Thickness
699-48-50	UN	UN	NP	NP	UN	UN	195
699-49-55B	530	12	518	98	420	27	137
699-49-57B	552	17	535	103	432	38	158
699-50-53	555	60	495	80	415	15	155
699-52-52	UN	UN	NP	NP	UN	UN	153
699-52-54	563	93	470	75	NP	NP	168
699-52-57	556	75	481	80	NP	NP	155
699-53-55	575	110	465	87	378	52	249
699-54-57	573	115	458	85	NP	NP	200
699-55-55	555	182	373	56	317	47	285
699-55-57	565	110	455	75	NP	NP	--
299-E28-26	687	60	627	200	427	50	310
299-E32-2	670	45	625	160	465	82	287
299-E33-12	620	20	600	170	430	42	232
299-E33-28	662	35	627	175	452	--	--
299-E33-29	673	55	618	160	458	25	240
299-E33-30	662	40	622	165	457	72	277
299-E33-31	644	55	589	150	439	45	250
299-E33-32	657	30	627	194	433	46	270
299-E33-33	640	38	602	182	420	32	252
299-E33-34	630	15	615	175	440	60	250
299-E33-39	620	8	612	162	450	59	229
299-E33-40	621	50	571	130	441	48	228
299-E33-41	651	35	616	195	421	18	248
299-E34-2	630	60	570	146	424	35	241

Table 3-4. Hanford Sequence Elevations and Thicknesses. (Sheet 2 of 2)

Borehole #	Hug T[1]	Hug TH[2]	Hs T[1]	Hs TH[2]	Hlg T[1]	Hlg TH[2]	Total Thickness
[1]Thickness of sequence, ft. [2]Elevation of top of sequence, ft AMSL. NP = Not present UN = Undifferentiated Hanford Source: Hoffman et al. (1992) Note: Information on 299-E33-38 borehole not provided in reference.							

Table 3-5. Summary of Grain Size Analyses of Selected Soil Samples[1] Collected within the 200-BP-1 Operable Unit and Immediate Vicinity. (sheet 1 of 3)

Crib No.	Permanent Well No.	Depth (m bls)	% silt/clay <0.074 mm	% sand 0.074-2 mm	% gravel >2 mm	% finer than 0.8 mm	Moisture Content, %
Surface Soil Samples (<15 ft depth)							
NA	299-E33-38	1.9	24	28	48	43	2.54
NA	299-E33-38	3.7	5	19	76	16	3.24
NA	299-E33-38	4.6	5	63	31	30	2.49
NA	299-E33-39	1.5	12	20	68	26	5.57
NA	299-E33-40	1.5	8	30	62	23	1.7
NA	299-E33-40	2.4	4	12	86	11	1.4
57A	299-E33-304	3.4	14	33	53	35	3.37
61A	299-E33-307	1.7	14	17	69	26	1.85
61A	299-E33-307	2.7	11	11	78	19	2.61
49B	299-E33-312	2.1	7	45	48	35	NT
49A	299-E33-302	0.9	6	22	72	21	NT
46B	299-E33-299	0.9	8	78	14	54	NT
45A	299-E33-318	0.9	24	62	14	79	NT
Infiltration Gravel Samples							
57A	299-E33-304	4.6	11	4	85	14	2.6
61A	299-E33-307	3.0	4	6	90	8	1.72
61A	299-E33-307	3.7	7	16	77	17	1.66
61A	299-E33-307	4.11	8	16	76	18	2.99
Subsurface Samples (>4.6 m (15 ft))							
NA	299-E33-38	15.1	4	38	58	NA	3.18
NA	299-E33-38	51.0	3	54	43	NA	2.6
NA	299-E33-38	57.1	8	92	-	NA	2.47
NA	299-E33-38	57.9	87	13	-	NA	21.53

Table 3-5. Summary of Grain Size Analyses of Soil Samples[1] Collected within the 200-BP-1 Operable Unit and Immediate Vicinity. (sheet 2 of 3)

Crib No.	Permanent Well No.	Depth (m bls)	% silt/clay <0.074 mm	% sand 0.074-2 mm	% gravel >2 mm	% finer than 0.8 mm	Moisture Content, %
NA	299-E33-38	58.2	7	34	59	NA	3.46
NA	299-E33-38	62.3	5	16	79	NA	2.13
NA	299-E33-38	70.1	4	15	81	NA	NT
NA	299-E33-38	71.9	6	76	18	NA	NT
NA	299-E33-39	6.3	9	73	18	NA	4.34
NA	299-E33-39	9.4	2	96	2	NA	2.38
NA	299-E33-39	24.5	5	77	18	NA	2.59
NA	299-E33-39	31.1	2	75	23	NA	2.11
NA	299-E33-39	37.0	4	83	13	NA	1.83
NA	299-E33-39	38.6	5	89	6	NA	2.1
NA	299-E33-39	53.7	5	86	9	NA	2.36
NA	299-E33-39	55.1	4	69	27	NA	3.23
NA	299-E33-39	58.3	7	56	37	NA	2.59
NA	299-E33-39	66.4	8	20	72	NA	NT
NA	299-E33-39	68.3	4	14	82	NA	NT
NA	299-E33-39	68.9	6	35	59	NA	NT
NA	299-E33-40	13.7	3	34	63	NA	8.58
NA	299-E33-40	16.5	3	90	7	NA	3.82
NA	299-E33-40	53.3	4	82	14	NA	3.34
NA	299-E33-40	59.4	23	74	3	NA	3.34
NA	299-E33-40	59.8	9	43	48	NA	NT
NA	299-E33-40	61.3	11	32	57	NA	1.93

Table 3-5. Summary of Grain Size Analyses of Soil Samples[1] Collected within the 200-BP-1 Operable Unit and Immediate Vicinity. (sheet 3 of 3)

Crib No.	Permanent Well No.	Depth (m bls)	% silt/clay <0.074 mm	% sand 0.074-2 mm	% gravel >2 mm	% finer than 0.8 mm	Moisture Content, %
NA	299-E33-40	66.1	6	17	77	NA	4.9
NA	299-E33-40	92.6	18	51	31	NA	NT
NA	299-E33-40	94.2	52	48	0	NA	NT
NA	299-E33-40	96.0	16	20	64	NA	NT

Source: Hoffman (1992)

[1] Samples collected at depths greater than 4.6 m (15 ft.) for the Task 2 and 4 boreholes are not included herein.

Table 3-6. Representative Hydraulic Properties of the Hanford Site Unconfined Aquifer.

Stratigraphic Interval	Hydraulic Conductivity (cm/s)	(ft/d)
Hanford formation	E - 01 - E+01	E+03 - E+04
Undifferentiated Hanford and middle Ringold unit	E - 02 - E+00	E+02 - E+04
Middle Ringold unit	E - 03 - E - 01	E+01 - E+03
Lower Ringold unit	E - 05 - E - 03	E - 01 - E+01
Adapted from Graham et al. (1981); Ringold units are as defined by Myers et al. (1979).		

Table 3-7. Transmissivity and Hydraulic Conductivity Values for the Uppermost Aquifer System[a].

Well Number	T ft^2/day (cm^2/s)	Equivalent K ft/day (cm/s)
299-E18-1	700 (8)	50 (.02)
299-E18-2	2000 (21)	150 (.05)
299-E18-3	3000 (32)	210 (.07)
299-E25-22	150000 (1620)	6200 (2.2)
299-E25-34	>250000 (2660)	19000 (6.7)
299-E25-35	>80000 (860)	6500 (2.4)
299-E27-8	>68000 (730)	6800 (2.4)
299-E27-9	35000 (370)	3500 (1.3)
299-E27-10	35000 (370)	3500 (1.3)
299-E28-27	>48000 (520)	4800 (1.7)
299-E32-4	>9500 (100)	950 (.3)
299-E33-28	>53000 (580)	5300 (1.8)
299-E33-29	>51000 (540)	5100 (1.8)
299-E33-30	>56000 (600)	5600 (2)
299-E34-2	114000 (1230)	11400 (4)
299-E34-3	14000 (150)	1400 (0.5)
699-31-53A	14900 (160)	125 (0.05)
699-31-53B	14200 (150)	120 (0.05)
699-33-56	21000 (230)	170 (.06)
699-36-61B	4200 (45)	25 (.009)
699-43-43	37000 (390)	2050 (0.7)
699-44-42	76000 (810)	4000 (1.4)
699-52-54	2000 (21)	400 (0.14)
699-52-57	120 (1)	30 (0.01)
699-53-55C	80000 (860)	3300 (1.16)
699-55-50C	600000 (6500)	25000 (8.8)
699-55-50D	400000 (4280)	8500 (3)
699-55-60A	64000 (690)	1600 (0.6)
699-55-60B	400000 (4280)	7300 (2.5)

[a] from Newcomer et al. (1992a).

Table 3-8. Transmissivity and Equivalent Hydraulic Conductivity Values for the Confined Aquifer[a].

Well No.	Transmissivity ft^2/day (cm^2/s)	Equivalent Hydraulic Conductivity ft/day (cm/s)	Geologic Unit
699-60-57	10,000 (110)	770 (0.3)	Basalt
299-E16-1	950 (10)	20 (0.007)	Elephant Mountain Interflow Zone
299-E26-8	230 (2.4)	20 (0.007)	Rattlesnake Ridge Interbed
299-E33-12	50 (0.6)	0.4 (0.0001)	Rattlesnake Ridge Interbed
699-42-40C	300 (3.1)	4 (0.001)	Rattlesnake Ridge Interbed
699-42-40C	7 (0.08)	–	Elephant Mountain Interflow Zone
699-47-42	470 (5)	7 (0.002)	Rattlesnake Ridge Interbed
699-47-50	1000 (10)	12 (0.005)	Rattlesnake Ridge Interbed
699-49-55B	780 (8.1)	15 (0.005)	Rattlesnake Ridge Interbed
699-49-57B	150 (1.6)	15 (0.005)	Rattlesnake Ridge Interbed
699-50-45	175 (1.8)	4 (0.001)	Rattlesnake Ridge Interbed
699-50-48B	280 (3)	8 (0.002)	Rattlesnake Ridge Interbed
699-51-46	110 (1.2)	2 (0.0007)	Rattlesnake Ridge Interbed
699-52-46A	135 (1.4)	3 (0.001)	Rattlesnake Ridge Interbed
699-52-48	35 (0.3)	1 (0.0003)	Rattlesnake Ridge Interbed
699-53-50	640 (6.9)	13 (0.005)	Rattlesnake Ridge Interbed
699-54-57	140 (1.5)	2 (0.0007)	Rattlesnake Ridge Interbed
699-56-53	690 (7.4)	10 (0.003)	Rattlesnake Ridge Interbed

[a]Information from Newcomer et al. (1992b).

Table 3-9. Confined/Unconfined Aquifer Well Clusters and Associated Barometric Efficiency and Vertical Hydraulic Gradient Data.

Well Cluster	Barometric Efficiency	Maximum Vertical Hydraulic Gradient	Direction of Vertical Flow Component (During Max. Vertical Hydraulic Gradient)
299-E33-07 (Uppermost aquifer system - Hanford fm.)	--	0.0050 ft/ft	↑
299-E33-40 (Rattlesnake Ridge aquifer)	25%		
699-49-55A (Uppermost aquifer system - Hanford fm.)	--	0.0023 ft/ft	↑
699-49-55B (Rattlesnake Ridge aquifer)	22%		
699-49-57A (Uppermost aquifer system - Hanford fm.)	--	0.0015 ft/ft	↑
699-49-57B (Rattlesnake Ridge aquifer)	19%		
699-50-53A (Uppermost aquifer system - Hanford fm.)	--	0.0037 ft/ft	↑
699-50-53B (Rattlesnake Ridge aquifer)	15%		
699-55-57 (Uppermost aquifer system - Hanford fm.)	--	0.0038 ft/ft	↑
699-54-57 (Rattlesnake Ridge aquifer)	15%		
699-53-55A (1) (Uppermost aquifer system - Hanford fm.)	--	0.0034 ft/ft	↑
699-53-55AP (Pomona Basalt Member)	23%		

Source: Connelly et al. (1992).
1 - Swanson (1992). Gradient measured 1/9/92.
-- = not applicable

Table 3-10. Plant Species of the 200 Area Plateau. (Sheet 1 of 5)

Common Name	Scientific Binomial	Occurrence
slender fescue	*Festuca octoflora*	Dry areas
asparagus	*Asparagus officinalis*	Ditch banks and moist areas
aster	*Machaeranthera canescens*	Disturbed dry areas
barnyard grass	*Echinochloa crusgallii*	Ditch banks and moist areas
bastard toad-flax	*Comandra umbellata*	Dry areas
leafy beggar-ticks	*Bidens rondosa*	Ditch banks and moist areas
annual bluegrass	*Poa annua*	Dry areas
big sagebrush	*Artemisia tridentata*	Undisturbed dry areas
blazing Star	*Mentzelia albicaulis*	Dry areas
bracken fern	*Pteridium aquilinum*	Ditch banks and moist areas
bristly foxtail	*Setaria lutesceus*	Ditch banks and moist areas
Douglas's brodiaea	*Brodiaea douglasii*	Dry areas
Howell's brodiaea	*Brodiaea howellii*	Dry areas
alkali bulrush	*Scirpus maritimus*	Ditches and ponds aquatic emergent
softstem bulrush	*Scirpus validus*	Ditches and ponds aquatic emergent
common cattail	*Typha latifolia*	Ditches and ponds aquatic emergent
hoary false yarrow	*Chaenactis douglasii*	Dry areas
cheatgrass	*Bromus tectorum*	Solid waste burial grounds Disturbed dry area
common mullein	*Verbascum thapsus*	Ditch banks and moist areas

Table 3-10. Plant Species of the 200 Area Plateau[a]. (Sheet 2 of 5)

Common Name	Scientific Binomial	Occurrence
cottonwood	*Populus deltoides*	Ditch banks and moist areas around ponds
matted cryptantha	*Cryptantha circumscissa*	Dry areas
winged cryptantha	*Cryptantha pterocarya*	Dry areas
cottonbatting cudweed	*Gnaphalium chilense*	Disturbed moist areas
showy Townsend-daisy	*Townsendia florifer*	Dry areas
dandelion	*Taraxacum officinale*	Ditch banks and moist areas
Munro's globe mallow	*Sphaeralcea munroana*	Dry areas
Gray's desertparsley	*Lomatium grayi*	Dry areas
desertparsley	*Lomatium* spp.	Dry areas
Carey's balsamroot	*Balsamorhiza careyana*	Dry area
winged dock	*Rumex venosus*	Old pond bottom, disturbed moist areas
dogbane	*Apocynum spp.*	Ditch banks and moist areas
common dogbane	*Apocynum androsaemifolium*	Old pond bottom, moist areas
spring whitlowgrass	*Draba verna*	Dry areas
eveningprimrose	*Oenothera pallida*	Dry areas
tarweed fiddleneck	*Amsinckia tesselata*	Disturbed dry areas
threadleaf fleabane	*Erigeron filifolius*	Dry areas
shaggy fleabane	*Erigeron pumilus*	Dry areas
Great Basin gilia	*Gilia leptomeria*	Dry areas
goldenrod	*Solidago* spp.	Old pond bottom and moist areas
green rabbitbrush	*Chrysothamnus nauseosus*	Solid waste burial ground, disturbed dry areas

Table 3-10. Plant Species of the 200 Area Plateau[a]. (Sheet 3 of 5)

Common Name	Scientific Binomial	Occurrence
grey rabbitbrush	*Chrysothamnus viscidiflorus*	Solid waste burial grounds, disturbed dry areas
hardstem bulrush	*Scirpus acutus*	Ditches, ponds, aquatic emergent
slender hawksbeard	*Crepis atrabarba*	Dry areas
smooth scouringrush	*Equisetum laevigatum*	Shoreline, wet areas
Dutch scourmgrush	*Equisetum hyemale*	Ditch banks and moist areas
horseweed	*Conyza canadensis*	Disturbed dry areas
Indian ricegrass	*Oryzopsis hymenoides*	Dry, sandy areas
Jimhill mustard	*Sisymbrium altissimum*	Solid waste burial grounds, disturbed dry areas
upland larkspur	*Delphinium nuttallianum*	Dry areas
white-daisy tidytips	*Laya glandulosa*	Dry areas
locoweed	*Astragulus* spp.	Dry areas
pink microsteris	*Microsteris gracilis*	Dry areas
alkali muhly	*Muhlenbergia asperifolia*	Ditch banks and moist areas
cutleaf mustard	*Thelypodium laciniatum*	Dry areas
needle-and-thread grass	*Stipa comata*	Sandy, dry areas
stinging nettle	*Urtica dioica*	Disturbed moist areas
common witchgrass	*Panicum capillare*	Ditch banks
peachleaf willow	*Salix amygdaloides*	Ditch banks, pond shoreline
sand beardtongue	*Penstemon accuminatus*	Dry areas
threadleaf scorpionweed	*Phacelia linearis*	Solid waste burial grounds, dry areas
longleaf phlox	*Phlox longifolia*	Dry areas
slimleaf goosefoot	*Chenopodium leptophyllum*	Dry areas

Table 3-10. Plant Species of the 200 Area Plateau[a]. (Sheet 4 of 5)

Common Name	Scientific Binomial	Occurrence
Indian wheat	*Plantago patagonica*	Moist areas
white cupseed	*Plectritis macrocera*	Moist areas
reed canarygrass	*Phalaris arundinacea*	Moist areas
Russian knapweed	*Centaurea repens*	Disturbed moist areas
Russian thistle	*Salsola kali*	Solid waste burial grounds, tank farms, dry areas
sand dropseed	*Sporobius cryptandrus*	Ditch banks, disturbed areas
coyote willow	*Salix exiqua*	Ditch banks, pond shoreline
bur ragweed	*Ambrosia acanthicarpa*	Solid waste burial grounds, disturbed dry areas
Sandberg's bluegrass	*Poa sandbergii*	Undisturbed dry areas
white sandverbena	*Abronia mellifera*	Dry areas
heartweed	*Polygonum persicaria*	Ditch banks and pond shoreline
water speedwell	*Veronica anagallis-aquatica*	Ditch banks, pond shoreline, moist areas
spiny hopsage	*Grayia spinosa*	Solid waste burial sites, dry areas
bottlebrush squirreltail	*Sitanion hystrix*	Dry areas
wester stickseed	*Lappula redowskii*	Disturbed dry areas
salt rattlepod	*Swainsona salsula*	Ditch banks, moist areas
white sweetclover	*Melilotus alba*	Ditch banks, disturbed mosit areas
western tansy mustard	*Descurainea pinnata*	Solid waste burial grounds, disturbed dry areas

Table 3-10. Plant Species of the 200 Area Plateau[a]. (Sheet 5 of 5)

Common Name	Scientific Binomial	Occurrence
Palouse thistle	*Cirsium brevifolium*	Moist areas
threesquare bulrush	*Scirpus americanus*	Ditches and ponds, aquatic emergent
rough wallflower	*Erysimum asperum*	Dry areas
watercress	*Rorippa nasturtium-aquaticum*	Ditches, running water
shrubby willowherb	*Epilobium suffruticosum*	Disturbed dry areas
prickly lettuce	*Lactuca serriola*	Ditch bank, disturbed moist areas
Yarrow	*Achillea millifolium*	Disturbed dry areas

[a]After Rogers and Rickard (1977).

Table 3-11. Bird species associated with the big sagebrush/Sandberg's bluegrass habitat of the 200 Area plateau[a]. (Sheet 1 of 2)

Common Name	Scientific Binomial	Status	
		Site Use[b]	Protection[c]
northern harrier	*Circus cyaneus*	B,Y	
Swainson's hawk	*Buteo swainsoni*	B	FC_3
red-tailed hawk	*Buteo jamaicensis*	B,Y	
rough-legged hawk	*Buteo lagopus*	W	
ferruginous hawk	*Buteo regalis*	B	FC_3, ST
American kestrel	*Falco sparverius*	B,Y	
prairie falcon	*Falco mexicanus*	B,Y	SM
long-billed curlew	*Numenius americanus*	B	FC3, SM
rock dove	*Columba livia*	B,Y	
mourning dove	*Zenaida macroura*	B,Y	
burrowing owl	*Athene cunicularia*	B	SC
long-eared owl	*Asio otus*	B,Y	
common nighthawk	*Chordeiles minor*	B	
northern flicker	*Colaptes auratus*	B,Y	
western kingbird	*Tyrannus verticalis*	B	
Say's phoebe	*Sayornis saya*	B	
horned lark	*Eremophila alpestris*	B,Y	
barn swallow	*Hirundo rustica*	B	
black-billed magpie	*Pica pica*	B,Y	
common raven	*Corvus corax*	B,Y	
sage thrasher	*Oreoscoptes montanus*	B	SC
American robin	*Turdus migratorius*	B,Y	
Townsend's solitaire	*Myadestes townsendii*	M	
loggerhead shrike	*Lanius ludovicianus*	B,Y	FC2
European starling	*Sturnus vulgaris*	B,Y	
western meadowlark	*Sturnella neglecta*	B,Y	
red-winged blackbird	*Agelaius phoeniceus*	B	
Brewer's blackbird	*Euphagus cyanocephalus*	B	

Table 3-11. Bird species associated with the big sagebrush/Sandberg's bluegrass habitat of the 200 Area plateau[a]. (Sheet 2 of 2)

Common Name	Scientific Binomial	Status	
		Site Use[b]	Protection[c]
northern oriole	*Icterus galbula*	B	
brown-headed cowbird	*Molothrus ater*	B	
vesper sparrow	*Pooecetes gramineus*	B	
Brewer's sparrow	*Spizella breweri*	B,Y	
white-crowned sparrow	*Zonotrichia leucophrys*	B,Y	
lark sparrow	*Chondestes grammacus*	B	
savannah sparrow	*Passerculus sandwichensis*	B	
chirping sparrow	*Spizella passerina*	M	
sage sparrow	*Amphispiza belli*	B	SC
house finch	*Carpodacus mexicanus*	B,Y	

[a] from Brandt and Rickard (1992) and Fitzner et al. (1992)
[b] B=breeding; Y=year-round resident; W=winter resident; M=seasonal migrant
[c] FCx=federal candidate, category x; ST=State threatened; SC=State candidate; SM=State monitor

Table 3-12. Mammal Species Associated with the Big Sagebrush/Sandberg's Bluegrass Habitat, Buildings, and Disturbed Areas of the 200 Area Plateau[a]. (Sheet 1 of 2)

Common Name	Scientific binomial	Protection[b]
Merriam's shrew	*Sorex merriami*	SC
vagrant shrew	*Sorex vagrans*	
little brown bat	*Myotis lucifugus*	
fringed bat	*Myotis thysanodes*	
California bat	*Myotis californicus*	
small-footed bat	*Myotis leibii*	
long-legged bat	*Myotis volans*	
long-eared bat	*Myotis evotis*	
hoary bat	*Lasiurus cinereus*	
silvery-haired bat	*Lasionycteris noctivagans*	
big brown bat	*Eptesicus fuscus*	
western pipistrelle	*Pipistrellus hesperus*	
pallid bat	*Antrozous pallidus*	
Pacific western big-eared bat	*Plecotus townsendii townsendii*	FC_2, SC
coyote	*Canis latrans*	
striped skunk	*Mephitis mephitis*	
long-tailed weasel	*Mustela frenata*	
badger	*Taxidea taxus*	
bobcat	*Lynx rufus*	
mule deer	*Odocoileus hemionus*	
Townsend's ground squirrel	*Spermophilus townsendii*	
northern pocket gopher	*Thomomys talpoides*	
Great Basin pocket mouse	*Perognathus parvus*	
bushy-tailed woodrat	*Neotoma cinerea*	
deer mouse	*Peromyscus maniculatus*	

Table 3-12. Mammal Species Associated with the Big Sagebrush/Sandberg's Bluegrass Habitat, Buildings, and Disturbed Areas of the 200 Area Plateau[a]. (Sheet 2 of 2)

Common Name	Scientific binomial	Protection[b]
western harvest mouse	*Reithrodontomys megalotis*	
northern grasshopper mouse	*Onychomys leucogaster*	
sagebrush vole	*Lagurus curtatus*	
Norway rat	*Rattus norvegicus*	
house mouse	*Mus musculus*	
porcupine	*Erethizon dorsatum*	
Nuttall's cottontail rabbit	*Sylvilagus nuttalli*	
black-tailed jackrabbit	*Lepus californicus*	

[a]after Hedlund and Rogers 1976, Rogers and Rickard 1977, Gano 1979, Marr et al. 1988.
[b]FC_x=federal candidate, category x; SC=State candidate.

4.0 NATURE AND EXTENT OF 200-BP-1 OPERABLE UNIT CONTAMINATION

This chapter provides a description of the nature and extent of contamination at the 200-BP-1 operable unit. The focus of Chapter 4 is on the empirically-determined chemical and contaminant characteristics of the operable unit. A description of the physical characteristics of the 200-BP-1 operable unit is provided in Chapter 3.

Section 4.1 describes information about the contaminant sources in the 200-BP-1 operable unit, including a summary of waste generating processes and estimated disposal quantities and chemical inventories at each waste management unit. Based on the review of this historic information, a list of anticipated contaminants is presented for each waste management unit. This list is provided for background information. Other waste streams in the operable unit vicinity are also briefly characterized.

Sections 4.2 through 4.4 consist of a data evaluation to identify contaminants of potential concern for crib soils and groundwater within the study area of the 200-BP-1 operable unit. The approach used in this evaluation is a step-wise screening process, based on guidelines for defining risk presented in EPA (1989a), Bleyler (1988a,b) and DOE-RL (1993), which narrows the list of the detected contaminants to those which represent actual contamination and which pose potential risk. The soil and groundwater contaminants of potential concern which result from this screening process are carried forward for use in the risk assessment performed in Chapter 6.

The extent of contamination discussion of this chapter focuses primarily on soil and groundwater media. The extent of contamination in other media (air, surface water and biota) is addressed primarily through the collection and use of existing data, and is discussed in section 4.2.3.

In Section 4.2, compounds detected in soil and groundwater are observed and compiled. Blank adjustments are performed on the reported results to eliminate detects which may have resulted from sampling equipment and/or shipment. Field and trip blank data are utilized in the blank adjustments to adjust the reported detected compounds accordingly. Sample detects which fall below calculated adjustment factors are considered as non-detects.

Next, the compounds which remain after the blank adjustment step are compared to established background concentrations to eliminate sample detects which represent naturally-occurring chemical constituents. Background concentrations (upper tolerance limits [UTLs]) are determined from published sources and sample data collected from uncontaminated areas. Exceedance of the UTLs constitutes evidence of contamination. Compounds that do not exceed the UTLs are dropped from further analysis.

The comparison to background is performed only for TAL parameters, bismuth, silicon, anions, gross alpha, gross beta, and the naturally occurring radionuclides (K-40, Ra-226, Th-228 and total uranium). Any detected (and blank adjusted) TCL organic, and radionuclide other than K-40, Ra-226, Th-228 and total uranium, are considered potential contamination because most of these compounds are not naturally occurring at detectable concentrations.

In Section 4.3, sample detects for the remaining compounds are compared to risk-based and regulatory criteria for each medium. Contaminants of potential concern are defined in Section 4.3 as those remaining compounds that are higher in concentration than the most stringent risk-based or regulatory concentration criteria for each respective medium. The most stringent criteria concentration is a conservative value for screening purposes and is not intended to represent cleanup criteria.

Section 4.4 presents a discussion of contaminant distribution and extent for each of the contaminants of potential concern identified in Section 4.3. Based on this discussion, some contaminants of potential concern are eliminated from further evaluation based on frequency of detection. Upper Confidence Limit (UCL) values are calculated and presented for the soil contaminants of potential concern. For groundwater, plume maps are presented to indicate the locations and extent of the contaminant plumes. Historical trends in contaminant concentrations are presented to highlight the temporal changes and significant trends in contaminant levels which have occurred at key wells throughout the study area.

4.1 CONTAMINANT SOURCES

The 200-BP-1 operable unit includes nine inactive cribs (known as the 216-B cribs) and four unplanned releases (UNs) (Figure 3-1). A tenth crib (216-B-61) was constructed, but there is no evidence that it was ever used or received any wastes (DOE-RL 1990a). The physical descriptions and operational characteristics of the cribs, and unplanned releases, are presented in Chapter 3. Table 3-1 identifies the cribs and UNs contained within the 200-BP-1 operable unit and summarizes their periods of operation (or dates of occurrence for UNs, if known) and waste sources. Appendix H provides additional information on the cribs and UNs associated with the 200-BP-1 operable unit including chemical inventories.

This subsection presents a review of the known and suspected chemical contaminants which have been associated with the waste management units at the 200-BP-1 operable unit, either through disposal activities, or through unplanned releases. Included is historic information about waste generating processes in the 200-BP-1 operable unit vicinity, and summaries of the known data regarding waste chemical inventories and disposal quantities at each waste management unit. The quantities of waste constituents disposed in the 200-BP-1 operable unit cribs are calculated estimates. There is uncertainty associated with these estimates, as well as with the identification of all constituents that were disposed of in the 200-BP-1 operable unit.

4.1.1 Waste Generating Processes

The cribs within the 200-BP-1 operable unit received liquid, mixed waste from two sources: low-level liquid waste from the U Plant uranium reclamation operations, and waste storage tank condensate from the adjacent 241-BY Tank Farm (Figure 3-1). Table 4-1 presents the estimated contaminant discharges and effluent volumes that were disposed of in the cribs.

4.1.1.1 TBP Process and Cribs 216-B-43 through -49. Waste sent to cribs 216-B-43 through -49 was generated in the tributyl phosphate (TBP) process in the 221-U Building. The TBP process was used for the recovery of uranium from wastes generated by the bismuth

phosphate ($BiPO_4$) process in the B Plant. Before implementing the TBP process, this waste had been stored in the 241-BY Tank Farm.

From 1952 to 1958, stored waste within the 241-BY tanks was transferred to the U Plant for uranium recovery. The sludge was dissolved in nitric acid, and then the uranium was extracted using TBP in a normal paraffin diluent. The TBP process wastes contained fission products, sulfate, and phosphate ions in aqueous nitric acid solution. The acid solution was made alkaline for transfer and storage in 241-BY Tank Farm. The TBP process wastes in the 241-BY tanks were treated with potassium ferrocyanide [$K_4Fe(CN)_6$] to precipitate cesium. The supernatant was decanted to cribs 216-B-43 through -49 (Anderson and Mudd 1979). Although the majority of cesium was effectively precipitated in the tanks, some cesium was left in the supernatant along with cyanide species and were discharged to the cribs. Chemical and radioactive constituents known and/or suspected to be present in the decanted supernatant included the following (DOE-RL 1992a and Stenner et al. 1988):

ferrocyanide	cesium-137	tritium (H-3)
nitrate	ruthenium-106	cobalt-60
phosphate	strontium-90	technetium-99
sodium	plutonium	
sulfate	alpha	
tributyl phosphate	beta	
paraffin hydrocarbons	uranium	

Table 4-1 presents the estimated disposal quantities at each crib for each of these constituents. These chemicals and radioactive constituents represent the anticipated contaminants associated with cribs 216-B-43 through -49.

4.1.1.2 In-Tank Solidification Process and Cribs 216-B-50 and -57. Waste sent to cribs 216-B-50 and -57 consisted of storage tank condensate from the in-tank solidification units nos. 1 and 2 (ITS nos. 1 and 2), respectively. In-tank solidification was accomplished by artificial in-tank heating using the ITS systems. These tanks were not self-boiling. Both units were located in the 241-BY Tank Farm (Figure 3-1). The ITS no. 1 unit startup occurred on March 19, 1965, and the ITS no. 2 unit started up on February 17, 1968. On August 24, 1971, the ITS no. 1 unit was converted from an evaporator to a cooler for ITS no. 2. Evaporates were collected and condensed. Condensate was discharged to either the 216-B-50 or the 216-B-57 crib. The 216-B-61 crib was constructed to receive ITS no. 1 unit condensate, but there is no documentation that it received any process wastes. Chemical and radioactive constituents known and/or suspected to be present in the storage tank condensate included the following (DOE-RL 1992a and Stenner et al. 1988):

ammonium carbonate	cesium-137	alpha	cobalt-60
ammonium nitrate	ruthenium-106	beta	technetium-99
nitrate	strontium-90	uranium	
sodium	plutonium	tritium (H-3)	

Table 4-1 presents the estimated disposal quantities at each crib for each of these constituents. These chemicals and radioactive constituents represent the anticipated contaminants associated with cribs 216-B-50 and -57.

4.1.2 Unplanned Releases

Four unplanned releases (UNs) have been identified (Figure 3-1) within the 200-BP-1 operable unit:

- UN-200-E-9
- UN-200-E-63
- UN-200-E-110
- UN-200-E-89.

Little information is available regarding the details of these UNs. Their known physical characteristics are discussed in Chapter 3. Additional information about each UN is included in Appendix H. This subsection is intended to summarize the available contaminant information for each UN.

Waste unit UN-200-9 involved approximately 41,635 L (11,000 gls) of TBP supernatant waste which leaked onto the ground from the 216-B crib flush tank. The spill occurred in an area directly north of the flush tank. Most of the wastes were removed to a site south of the 216-B-43 crib and were covered with 0.6 m (2 ft) of clean soil. The contamination left near the flush tank was covered with 3 m (10 ft) of clean soil (DOE-RL 1992a). Anticipated contaminants for this unplanned release are essentially identical to those identified above for the 216-B-43 through -49 cribs.

UN-200-E-63 was an unplanned mixed waste release that occurred June 4, 1981. The release is described as tumbleweeds becoming contaminated by uptake of radionuclides from the BC crib and trench. This vegetation was then blown and contaminated the ground surface. The contaminated vegetation was removed and a weed control program was initiated to control future growth of tumbleweeds.

Waste unit UN-200-110 involved first-cycle waste from the 112-BY tank in the 241-BY Tank Farm and impacted a crescent-shaped ground area of approximately 2,320 m^2 (25,000 ft^2) around the 112-BY valve pit. The area was contaminated to 22 R/h. It is possible, although currently unknown, that the release flowed into the 200-BP-1 operable unit. Anticipated contaminants for this unplanned release are expected to be similar to those disposed in the 216-B-43 through -49 cribs.

UN-200-E-89 consisted of surface radiation that exceeded allowable levels and which was documented in a series of surveillance reports. The majority of UN-200-E-89 (approximately 4.3 ha [10.6 ac]) was within the 200-BP-1 operable unit. The remainder (approximately 0.85 ha [2.1 ac]) was within the 200-BP-7 operable unit to the south (Hayward 1992). The location of the UN within the 200-BP-1 operable unit is shown in Figure 3-1. Interim stabilization actions were conducted at the UN consisting of a combination of scraping and re-placement of surface contaminated soils followed by covering (stabilization) with clean soil and rock, and covering (stabilization) of soils in-place. Details of this action are discussed in Chapter 3.

The source of the contamination was suspected to be the manhole and risers near the 216-B cribs, and the BX/BY Tank Farms (Hayward 1992). As such, the anticipated contaminants include the same compounds which were mentioned above for the 216-B-43 through -49 cribs, and for cribs 216-B-50 and -57.

4.1.3 Other Potential Contaminant Sources in the 200-BP-1 Operable Unit Vicinity

The 200-BP-1 operable unit is bordered by operable units 200-BP-4 to the east, 200-BP-7 and 200-BP-3 to the south, and the 200-BP-10 operable unit to the west (Figure 3-2). The 600 Area, which includes all areas at the Hanford Site not located in the other designated Areas (i.e., 100, 200, 300, 400, and 1100), borders the operable unit to the north.

Operable units adjacent to 200-BP-1 contain waste storage, waste burial, or waste infiltration systems. The 200-BP-4 and 200-BP-3 operable units contain cribs. The 200-BP-7 operable unit contains the 241-BY, 241-BX, and 241-B Tank Farms. The 200-BP-10 operable unit contains burial grounds for solid waste disposal. The 600 Area immediately to the north of the 200-BP-1 operable unit does not contain process facilities or waste disposal systems.

The waste disposal cribs and waste storage tanks in adjacent operable units received or contain many of the same constituents as the cribs in the 200-BP-1 operable unit. Cribs in adjacent operable units were designed for underground infiltration of waste water effluents. Leaks have occurred from single-shell storage tanks in the 200-BP-7 operable unit, resulting in release of wastes containing higher concentrations of radionuclides than were normally disposed of in the cribs (WHC 1988).

In addition to the operable units immediately adjacent to the 200-BP-1 operable unit, other major potential contaminant sources include B-pond effluent infiltration to groundwater, reverse well 216-B-5 discharges to groundwater and possible stack emission fallout associated with processing plants in the 200 East Area.

4.2 200-BP-1 OPERABLE UNIT CONTAMINATION

This section consists of a data evaluation performed to define the compounds detected in 200-BP-1 soil and groundwater samples that constitute actual contamination. It is the first portion of a step-wise screening process which results in the identification of contaminants of potential concern (Section 4.3) for use in the risk assessment performed in Chapter 6. As such, the detected compounds are subjected to the following two initial screening steps:

- Case blank adjustments
- Comparison to soil and groundwater background concentrations.

Case blank adjustments are performed on the detected compounds to eliminate those detects which may have resulted from sampling or analytical equipment and/or shipment. Validated field and trip blank data are utilized to adjust the reported detected compounds. Sample detects which fall below case blank adjustment factors are adjusted to non-detects.

The blank adjustments conducted here remove systematic detect bias for media analysis conducted by a given laboratory during a sampling event, or case (Bleyler 1988a,b). A case is defined for soil as all soil samples delivered to a particular laboratory over the entire course of this RI investigation, and for groundwater as all samples delivered to a particular laboratory for each quarterly sampling round. Each sample case is made up of one or more sample delivery groups (SDGs). This case blank adjustment procedure is separate from, and in addition to, the SDG blank adjustments which were performed

during the data validation process. SDG blank adjustments were performed for each SDG (during data validation) using blank data obtained from and associated with each SDG only. The case blank adjustments are performed over an entire sample case utilizing all validated blank data from the sample case.

Case blank adjustments were not performed for those cases where no validated field or trip blank data was obtained. In these instances, all detected compounds were carried forward to the next step in the screening process (background comparisons).

Following the case blank adjustments, background comparisons are performed on the detected and case blank adjusted compounds to eliminate those sample detects which represent naturally-occurring chemical concentrations. The background screening is performed only for TAL parameters, bismuth, silicon, inorganic anions naturally occurring radionuclides (K-40, Ra-226, Th-228 and total uranium), gross alpha and gross beta. Any detected TCL organic and radionuclides other than K-40, Ra-226, Th-228 and total uranium are considered potential contamination because most of these compounds are not naturally occurring at concentrations above analytical detection levels.

Background concentrations (UTLs) are obtained from operable unit-specific background soil samples, and from the reports on Hanford Site Soil Background (DOE-RL 1992b) and Hanford Site Groundwater Background (DOE-RL 1992c). In this RI, the published values are used primarily, and are supplemented when necessary with calculated UTLs based on the background soil sampling. All groundwater UTLs are obtained from the published values.

Potential contamination of each media is evaluated by comparing the detected and blank adjusted parameter concentrations with background UTLs. Exceedance of the UTLs constitutes evidence of contamination. All compounds which remain after the case blank adjustments and comparison to background are carried forward into the risk-based screening of Section 4.3. Compounds which are screened out are not considered further in this RI.

4.2.1 Soils

Surface and subsurface soil samples were obtained from boreholes constructed during Tasks 2, 4, and 6 and from surface samples obtained during Task 3. Task 2 and 4 boreholes were drilled through the 216-B cribs while Task 6 boreholes were located in areas adjacent to the cribs, and in the 600 Area. The locations of the Task 2 and 4 boreholes are indicated in Figure 2-1. Table 2-2 summarizes relevant borehole completion data. Task 6 borehole locations are depicted in Figure 2-4 and summarized in Table 2-4. Surface soil sampling locations are shown in Figure 2-3. A summary of the sampling scheme for the various tasks is presented in Table 2-1.

Soil samples from Task 2 and 4 were analyzed for TCL organics, TAL metals, bismuth, major anions, cyanide (total, free and complexed) and radionuclides expected to be present in the waste effluent disposed in the cribs. Soil samples from Task 3 were analyzed for major anions, bismuth, selenium, cyanide (total) radionuclides. Task 6 soil samples were analyzed for TOC, major anions, bismuth, selenium, cyanide (total and free) and radionuclides. The Task 6 soil sample results were used only for the calculation of background UTLs and were not used in the screening process. Soil sample analyses were

performed at two facilities: an on-site lab operated by PNL and an off-site facility operated by Weston. The results of the Tasks 2 and 4, Task 3 and Task 6 analytical work are represented in Appendices A, C, and D, respectively.

4.2.1.1 Case Blank Adjustments. Only samples analyzed by Weston Labs were blank adjusted. PNL data was not blank adjusted as there were no trip blanks sent to PNL, except for one VOA sample with no detects. PNL analyses of crib soils were for high activity samples requiring "hot cell" analysis. Blank samples would be extremely expensive to analyze within "hot cells". The Task 3 surface soil samples were not blank adjusted because there were no blank sample results available.

Blank adjustments were performed for TCL parameters and radionuclides only. Blank adjustment was not performed on TAL parameters and anions because the trip blanks consisted of water samples as required by EPA. These are not appropriate as indicators of TAL compound contamination in soils because of the different sample matrices and analytical preparation procedures.

The compounds detected in the blanks and their corresponding maximum blank concentrations are presented in Table 4-2. In accordance with Bleyler (1988a,b), the maximum value detected in a blank sample is multiplied by a factor of either 5 or 10, depending on the parameter. This adjustment value is then compared with the reported sample results. If the reported value does not exceed the adjustment criterion, the sample result is regarded as undetected. The number of sample blank adjustments which were made are listed in Table 4-2. TCL parameter concentrations in water (ppb) were used directly to adjust soil concentrations in ppb. Radionuclide concentrations in pCi/L were converted to pCi/ml in order to adjust soil concentrations in pCi/g. These adjustments have been applied to the data set for the Tasks 2 and 4 sampling events summarized in Appendix A.

4.2.1.2 Background Soil Quality. The comparison to background is performed only for the TAL parameters, inorganic anions and the radionuclides gross alpha, gross beta, K-40, total uranium, thorium-228 and radium-226. Any detect for a TCL organic or radionuclide other than K-40, total uranium, thorium-228 and radium-226 is considered evidence of contamination as most of these compounds are not naturally occurring at detectable concentrations.

This approach is conservative because general anthropogenic sources of organics and fission product radionuclides not attributable to Hanford operations were not included as background. Atmospheric fission product fallout from anthropogenic sources not attributable to Hanford operations probably has occurred and continues to occur on the Hanford Site. These sources are anticipated to be minor, but have not been included as background.

Background soil quality (background UTLs) is summarized in Table 4-3. In the table, soil UTLs from two sources are indicated. These two sources include:

- project specific background soil samples from uncontaminated areas
- published values from Hanford Site Soil Background (DOE-RL 1992b).

For determination of 200-BP-1 operable unit extent of contamination, background concentrations presented in DOE-RL (1993a) shall apply, except in the case of constituents

for which no values were published. Background UTLs calculated from the operable unit-specific soil samples shall be used in these instances. For the TAL parameters bismuth and the naturally occurring radionuclides, exceedance of the background UTLs is regarded as evidence of contamination.

The operable unit-specific background soil samples were obtained from three boreholes in and near the 200 East Area (Figures 2-1 and 2-4) (699-52-57, 699-55-55 and 299-E33-307). Borehole 299-E33-307 is located in crib 216-B-61, in the northwestern portion of the 200-BP-1 operable unit away from the main crib area. Crib 216-B-61 was reportedly never used for waste disposal. Boreholes 699-52-57 and 699-55-55 are within the 600 Area, approximately 1.1 and 1.9 km (0.7 and 1.2 mi) north of the 200-BP-1 operable unit, respectively. No discernable concentration differences were noted between locations; therefore, all the background samples which were taken above water table were considered together to provide a description of the operable unit-specific background conditions.

Background samples were not differentiated based on surface or subsurface stratification due to the small number of samples. Saturated samples were excluded because of the potential for lateral contamination via groundwater movement.

Background soil quality from the operable unit-specific background samples were analyzed by means of the one-sided, upper tolerance limit (UTL) for the 95th percentile (alpha = 0.05) for the distribution of each parameter. A normal distribution is conservatively assumed, as data are of insufficient quantity to justify otherwise. The method for the calculation of the background UTLs is provided in EPA (1989b). As seen in Table 4-3, the calculated background UTLs and mean are comparable to most background soil concentrations presented in Hanford Site Soil Background (DOE-RL 1993a). Again, the Hanford Site Soil Background (DOE-RL 1993a) values were used, except in the case were parameter background values were not published then site specific values were used.

All inorganic concentrations are reported in terms of mg/kg (parts per million), organic results are reported in micrograms/kg (parts per billion), while all radionuclide concentrations are reported in pCi/g (picocuries per gram). These are the units reported by the analytical laboratories. If a parameter was not detected in a given background sample, one-half the SQL was used as a surrogate value in the statistical calculation. If a given parameter was never detected in a respective set of background samples, the highest reported SQL for the parameter is substituted for the UTL, as noted in Table 4-3.

Exceedance of the background UTL is regarded as evidence of contamination. Compounds which were not detected in any soil sample at reported concentrations above the background UTLs are dropped from further consideration in this RI. Table 4-4 summarizes the results of the background screening for the TAL parameters, inorganic anions and naturally occurring radionuclides. As seen in the table, 15 inorganic compounds fell below the background levels and will not be considered further. The compounds which were detected at levels above the background UTLs and all detected organics and fission product radionuclides will be carried forward into the risk-based screening presented in Section 4.3.

4.2.2 Groundwater

The local unconfined and confined groundwater aquifers were sampled under Task 7 (Groundwater Sampling and Analysis) of the 200-BP-1 operable unit Phase I RI (DOE-RL 1990a). Samples were collected quarterly beginning in January 1991 from previously existing and recently installed monitoring wells for the 200-BP-1 operable unit RI. A total of five quarters of data have been collected and are included for consideration in this report. Sampling for the full suite of TCL organic compounds was conducted only for the first quarter sampled (winter 1991). For the four succeeding quarters TCL parameters were analyzed for only trip blanks. The locations of wells which were sampled are shown in Figure 2-5. A total of from 36 to 44 monitoring well samples were obtained for chemical analysis from each round of the groundwater sampling.

The analyses conducted on groundwater samples for each quarterly period include: TAL metals plus bismuth, silicon, tin and strontium, total cyanide, anions and radionuclides expected to be present in significant quantity in wastes disposed within the 200-BP-1 cribs. Analyses for TCL organic compounds were performed only on the first quarter samples in order to confirm that only low contaminant levels are present in groundwater, as was suspected from historic data. Free cyanide analyses were performed on all samples collected for the first two sampling events and on selected samples thereafter. The results of the groundwater chemical analyses are included as Appendix E.

4.2.2.1 Case Blank Adjustments. All validated trip blank data as of 11/12/92 were used for blank adjustment in accordance with Bleyler (1988a,b). Blank adjustments were performed for all target analytes (TAL and TCL parameters, and radionuclides). Only samples analyzed by Weston Laboratory were blank adjusted. PNL received samples for cyanide analysis only and none of the trip blanks had any detects.

The compounds detected in the blanks and their corresponding maximum blank concentrations are presented in Table 4-5. In accordance with Bleyler (1988a,b), the maximum value detected in a blank sample is multiplied by a factor of either 5 or 10, depending on the parameter. This adjustment value is then compared with the reported sample results. If the reported value does not exceed the adjustment criterion, the sample result is regarded as undetected. The number of sample blank adjustments which were made to groundwater samples are listed in Table 4-5. These adjustments have been applied to the data set summarized in Appendix D.

4.2.2.2 Background Groundwater Quality. As with the case of soil, the comparison to background was performed only for the TAL parameters, bismuth, silicon, tin, strontium, inorganic anions, the naturally occurring radionuclides (K-40, total uranium, thorium-228, Ra-226), gross alpha and gross beta. Any detected TCL organics or radionuclides other than K-40, total uranium, thorium-228 and Ra-226 are assumed to represent evidence of potential contamination as most of these compounds are not naturally occurring. This is a conservative approach because general anthropogenic sources of organics and fission product radionuclides that are not attributable to Hanford operations were not included as background.

Background groundwater concentrations (UTLs) for detected target parameters were not calculated from operable-unit specific samples. Upgradient and downgradient wells are likely to have been impacted from either the 200-BP-1 operable unit or from other sources in the vicinity of the 200 East Area. Therefore, all background concentrations for

groundwater are obtained from the report on Hanford Site Groundwater Background (DOE-RL 1992b). The values obtained for background groundwater quality (background UTLs) are summarized in Table 4-3. Unfortunately, background concentrations for naturally occurring radionuclides K-40 and thorium-228 were not determined in the Hanford Site Groundwater Background Study (DOE-RL 1992b).

Compounds which were not detected in any groundwater sample at reported concentrations above the background UTLs are dropped from further consideration in this RI. Table 4-6 summarizes the results of the background screening. As seen in the table, two constituents fell below the background levels and will not be considered further.

Several Hanford Site background concentrations are reported for manganese, iron, zinc, fluoride, chloride, gross alpha and gross beta (DOE-RL 1992b). The different values are listed in Table 4-3 as high, medium or low and may represent different concentration groupings within the unconfined aquifer. If a parameter was below any of the reported (high, medium or low) values it was considered to be below background and was dropped from further consideration. The compounds which are detected at levels above the highest background UTLs are carried forward into the risk-based screening presented in Section 4.3.

4.2.3 Other Media

Chapter 4 presents data describing the extent of 200-BP-1 operable unit contamination for soil and groundwater media only. Other media which may provide significant pathways for contaminant exposures, and which may need to be characterized and considered in the risk assessment, include air, surface water, and biota. Characterization of the nature and extent of contamination for these media is addressed in the following sections.

4.2.3.1 Air. In Chapter 5, predictive estimates of air quality are made for the 200-BP-1 operable unit vicinity through the use of a Fugitive Dust Model (Winges 1991). These calculations utilize data consisting of predominant weather conditions, and physical and chemical characteristics of the surface soils in the vicinity of the 200-BP-1 operable unit. The results of the Fugitive Dust Model analyses are carried forward into Chapter 6 for use in the Risk Assessment. Selected existing air quality data were also obtained to confirm the general validity and representativeness of the calculated results and to provide supplementary data where necessary. These data were gathered from the routine air quality monitoring program which is conducted at the Hanford Site and has been reported in Woodruff and Hanf (1992) and Schmidt et al. (1992). Discussion and use of this data are included in Section 6.1.

4.2.3.2 Surface Water. As discussed in Chapter 3, there is no surface water located on the 200-BP-1 operable unit. Nearby surface water bodies include West Lake, the Columbia River and B pond. The major pathway for impacts to West Lake and the Columbia River involves the groundwater pathway. An assessment of this pathway is not included in the scope of this RI but will be addressed in the 200 East Groundwater Aggregate Area Study. The B pond consists of a separate waste operable unit, and as such, will not be addressed in this RI. Therefore, no data will be presented in this RI describing the extent of contamination in surface water at, or near, the 200-BP-1 operable unit.

4.2.3.3 Biota. No biotic uptake sampling was performed in the 200-BP-1 operable unit as part of this RI. For the purposes of the risk assessment, contaminant levels in biota have been estimated using standard procedures and methods, as presented in Chapter 5.

4.3 RISK-BASED SCREENING

To focus the list of contaminants exceeding background to those with the greatest likelihood of dominating the overall risk at a waste management unit, a risk-based screening process is used to identify the contaminants of potential concern. The risk-based screening process is conducted for each waste management unit (or group of units if appropriate) as discussed in the *Hanford Site Baseline Risk Assessment Methodology (HSBRAM)* (DOE-RL 1993b). The procedure involves the calculation of risk-based benchmark concentrations against which the maximum detected concentration of a contaminant are compared. Risk-based benchmark concentrations are soil or groundwater concentrations that correspond to a specific hazard quotient (HQ - Section 6.2.3.1) or lifetime incremental cancer risk (LICR - Section 6.2.3.2) using defined exposure assumptions, as discussed below. If the maximum concentration detected for a contaminant does not exceed the risk-based benchmark concentrations for that contaminant, it can be eliminated from further consideration in the risk assessment. The screening process provides a high degree of confidence that these eliminated contaminants pose only an insignificant risk to human health or the environment.

Exceedance of a conservative risk-based benchmark concentration does not necessarily establish the existence of a significant risk. At this point in the overall analysis, it simply indicates the need to retain the contaminant for further evaluation in the risk assessment. As a supplement to the risk-based screening, potential contaminant-specific soil cleanup regulations are also compared to the maximum detected concentrations. Any contaminants exceeding potential regulatory cleanup guidelines are also retained for further evaluation in the risk assessment.

As recommended in the HSBRAM (DOE-RL 1993b), all preliminary risk-based benchmark concentrations are calculated using on-site residential exposure assumptions. On-site residential exposure assumptions are utilized for preliminary screening because they are more conservative than industrial exposure assumptions. In Chapter 6 of this RI report, an on-site industrial scenario is used to estimate contaminant intakes. Consequently, contaminants that may represent a significant risk (concentrations above risk-based benchmark concentrations) are retained for further evaluation in the baseline risk assessment.

The calculation of the preliminary risk-based benchmark concentrations considers both noncarcinogenic effects (i.e., systemic toxicity) and carcinogenic effects. As recommended in the HSBRAM (DOE-RL 1993b), risk-based benchmark concentrations are calculated for soil concentrations that would be equivalent to exposures at an HQ of 0.1 for contaminants with noncarcinogenic effects. A concentration equivalent to an LICR of 1E-07 is used for contaminants with carcinogenic effects. Equations for calculating risk-based benchmark concentrations are based on standard EPA intake equations. Exposure parameters are provided in Table 4-7 and are discussed more extensively in Appendix A of the HSBRAM (DOE-RL 1993b).

For this RI, preliminary risk-based screening is conducted for both the soil and groundwater exposure scenarios. The specific exposure pathways, exposure parameters and risk-based screening results are discussed in the sub-sections below for both soils and groundwater respectively.

4.3.1 Soils -Risk-based Screening Methodology

4.3.1.1 Risk-based Screening Approach and Assumptions. Due to the nature of crib construction and contaminant disposal at the 200-BP-1 operable unit, the subsurface matrix is composed of two distinct zones. The first zone is comprised of backfill over the cribs and mounded surface soils from the UN-200-E-89 stabilization. The second zone is comprised of infiltration gravels on top of, presumably, in-situ soils. The infiltration gravels are generally located 3.4 to 6.1 m (11 to 20 ft) below the ground surface. Due to the differences in matrix and contaminant levels with depth, risk-based screening was conducted separately for two different zones. The first zone, near surface soils, is defined as soil between the ground surface and 4.6 m (15 ft) below the ground surface. It is comprised primarily of stabilized surface soils (UN-200-E-89) and crib backfill with 0.3 to 1.2 m (1 to 4 ft) of gravel at the bottom of the zone. The second zone, which starts in the crib infiltration gravels, is defined as everything below 4.6 m (15 ft). The cut off of 4.6 m (15 ft) is based on MTCA guidelines, the extent of contamination within the cribs, and the depths at which the soil samples were taken. Analysis of the soil sampling data show the contamination to be greatest below 4.6 m (15 ft) and soil samples were generally taken from 2.7 to 3.6 m (9-12 ft) and 5.5 to 6.4 m (18-21 ft).

For both the near surface soil and subsurface infiltration gravels/soils, three soil exposure pathways are used in calculating preliminary risk-based benchmark concentrations, as recommended in the HSBRAM (DOE-RL 1993b). These exposure pathways are soil ingestion, inhalation of fugitive dust, and external exposure to radioactivity. The exposure assumptions for these pathways are discussed in Section 4.3.1.2 and summarized in Table 4-7.

For the purposes of this report, several assumptions are used with respect to the detected compounds:

- All detected polychlorinated biphenyls (PCBs) are combined and evaluated as total PCBs. Therefore, the values for PCBs in subsequent tables are the sums of the concentrations provided for the individual aroclors that had at least one detect (e.g., Aroclor 1232, and 1254).

- The maximum concentration analyzed within each matrix for each parameter is used to provide a conservative estimate of the contaminant concentration.

- All chromium is assumed to be chromium(VI), which has the lowest chronic toxic reference dose (RfD) or carcinogenic slope factor (SF) of the valence states of chromium. Insufficient data are available to ascertain speciation of chromium at the waste management units at this time. This assumption provides a conservative evaluation of the potential toxicity associated with chromium present in soil at the 200-BP-1 operable unit waste management systems.

- All cyanide is evaluated using the toxicity values for free cyanide (uncomplexed or easily dissociable), since this is the only form of cyanide with a toxicity factor. This is a conservative approach since analysis was performed for both total and free cyanide on all samples and for complexed cyanide on selected samples.

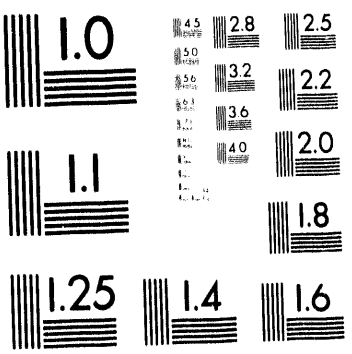
Another assumption is made that the form of nickel present at the site is not carcinogenic. The only form of nickel that has been determined to be carcinogenic in both humans and animals is nickel subsulfate via inhalation. Nickel refinery dust (about 50% nickel subsulfate) has been found to be carcinogenic in occupational workers who were exposed to the substance at pyrometallurgical sulfide nickel matte refineries via inhalation (SRC 1991a). Nickel refinery dust is usually generated during high temperature operations at manufacturing plants and is not the typical form of nickel found at other types of industrial sites. Since there is no pyrolytic activity at the site that will generate nickel refinery dust, all nickel detected at the site is not considered to be carcinogenic via inhalation. Evidence that nickel and its compounds are carcinogenic by the oral route have not been conclusively demonstrated in experimental animal studies (SRC 1991a). In addition, the predominant forms of nickel that are found in the environment (i.e., soil, water, etc.) are water-insoluble and are generally not bioavailable for most plant and animal species (SRC 1991a). Thus, nickel is evaluated only for its noncarcinogenic toxicity

Several contaminants which exceeded 200-BP-1 operable-unit-specific background soil concentration have been eliminated from further evaluation in this risk assessment using procedures recommended in *Risk Assessment Guidance for Superfund (RAGS)* (EPA 1989a), the EPA Region 10 (EPA-10) *Supplemental Risk Assessment Guidance for Superfund* (EPA-10 1991), and the HSBRAM (DOE-RL 1993b). These contaminants are magnesium, potassium and sodium. The maximum detected values for potassium and sodium are below the geometric mean of 17,000 and 10,200 mg/kg, respectively, for the Western United States (Shacklette et al. 1971). While the maximum detected value for magnesium is above the geometric mean for the Western U.S., ([7,800] Shacklette et al. 1971), magnesium has a low potential for toxicity under environmental conditions and is an essential human nutrient. Aluminum does not have an EPA-endorsed toxicity and is generally considered harmful only in relation to medical therapy (Amdur et al. 1991). Therefore, aluminum is eliminated from further evaluation.

Phosphate and bismuth are also eliminated from further consideration. Both phosphate and bismuth are generally not considered harmful to human health through the potential exposures to soil that would occur at this operable unit. Toxicity values are not provided by EPA to evaluate these substances. Most bismuth compounds are insoluble and are poorly absorbed either from the gastrointestinal tract or when applied to the skin. The toxicological hazards of bismuth salts are generally associated with their use in medical therapy and not from exposure to low levels that are usually present in environmental media (Amdur et al., 1991).

In the case of phosphates, the principal problem associated with these salts are not directly related to human health. Rather, the major concern for phosphates in the environment is the potential adverse impact to water quality (Amdur et al., 1991).

Lead does not have any published numerical toxicity values such as an RfD or SF that can be used to calculate risk-based benchmark concentrations. However, in the case of direct contact to lead in affected soils, the EPA (1989c) has indicated that a concentration


1.0
1.1
1.25
1.4
1.6
1.8
2.0
2.2
2.5
2.8
3.2
3.6
4.0
4.5
5.0
5.6

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range of 500-1,000 mg/kg in soil is considered protective of sensitive subpopulations (e.g., children) even at sites where the current or predicted land use is residential. Current MTCA cleanup guidelines are more stringent at 250 mg/kg in soil. The potential human receptors to soil at this waste unit are industrial workers and not children who are generally more sensitive to the adverse effects of oral lead exposure. However, for the sake of conservatism, lead is eliminated from further consideration at sites where the maximum detected concentrations are within or less than the most conservative standard of 250 mg/kg.

Additionally, radionuclide testing included analysis of gross alpha and gross beta concentrations within the soil. These measurements indicate the presence and overall magnitude of radionuclide concentrations within the soils. However, toxicity information for radionuclides is isotope specific, therefore measurements of gross alpha and gross beta cannot be evaluated in this risk assessment. Numerous radionuclides are present in the soil in the 200-BP-1 operable unit, all of which would contribute varying levels of alpha and beta radiation. Radionuclides anticipated to be present from operations at the site were analyzed. Daughter products can be estimated based on the concentration of the parent.

Table 4-8 lists the contaminants carried through this process for near surface soils and subsurface infiltration gravels/soils. For each contaminant the maximum concentration detected, the total number of detects and the total number of samples analyzed for that parameter are included in Table 4-8, to give a qualitative perspective on significance.

4.3.1.2 Risk-based Screening Calculations. The risk-based screening calculations are presented in the following sub-sections for the three exposure pathways, soil ingestion, inhalation of fugitive dust and external exposure to radionuclides. Example calculations are provided in Appendix I.

4.3.1.2.1 Soil Ingestion. All soil ingestion risk-based benchmark concentrations are derived using on-site residential exposure assumptions from the HSBRAM (DOE-RL 1993b). The residential exposure assumptions of the HSBRAM are based primarily on EPA-10 (1991) and the Washington State Model Toxics Control Act Cleanup Regulations (MTCA, Washington Administrative Code [WAC] 173-340). For carcinogens, the exposure assumptions are based on a child and an adult exposure. The child exposure factors are a body weight of 16 kg (35.2 lbs), a daily soil ingestion rate of 200 mg/d, an exposure frequency of 365 d/yr, and an exposure duration of 6 years. The adult exposure factors are a body weight of 70 kg (154 lb), a daily soil ingestion rate of 100 mg/d, an exposure frequency of 365 d/yr, and an exposure duration of 24 years. For oral soil ingestion of contaminants with noncarcinogenic effects, the exposure assumptions are based only on exposures for a child. The exposure factors, therefore, are a body weight of 16 kg (35 lb), a soil ingestion rate of 200 mg/d, an exposure frequency of 365 d/yr, and an exposure duration of 6 years. A conversion factor of 1E+06 mg/kg is used.

For carcinogenic non-radioactive contaminants, the screening equation is:

$$C = \frac{TR \times BW \times AT \times CF}{SF \times IR \times EF \times ED} \quad \textbf{4-4}$$

where:

C = risk-based benchmark concentration (mg/kg)
TR = target excess individual lifetime cancer risk (1E-07)
BW = body weight (adult: 70 kg; child: 16 kg)
AT = averaging time (365 d/yr x 70 yr)
CF = conversion factor (1E+06 mg/kg)
SF = contaminant-specific slope factor $(mg/kg\text{-}d)^{-1}$
IR = intake rate (adult: 100 mg/d; child: 200 mg/d)
EF = exposure frequency (365 d/yr)
ED = exposure duration (adult: 24 yr; child: 6 yr)

For noncarcinogenic effects, the equation is:

$$C = \frac{THQ \times RfD \times BW \times AT \times CF}{IR \times EF \times ED} \quad \textbf{4-5}$$

where:

C = risk-based benchmark concentration (mg/kg)
THQ = target hazard quotient (0.1)
RfD = contaminant-specific chronic reference dose (mg/kg-d)
BW = body weight (16 kg)
AT = averaging time (365 d/yr x 6 yr)
CF = conversion factor (1E+06 mg/kg)
IR = intake rate (200 mg/d)
EF = exposure frequency (365 d/yr)
ED = exposure duration (6 yr)

For radioactive contaminants, the general equation to calculate risk-based benchmark concentrations is:

$$C = \frac{TR \times CF}{SF \times IR \times EF \times ED} \quad \textbf{4-6}$$

where:

C = risk-based benchmark concentration (pCi/g)
TR = target excess individual lifetime cancer risk (1E-07)
CF = conversion factor (1E+03 mg/g)
SF = radionuclide-specific slope factor $[(pCi)^{-1}]$

IR = intake rate (adult: 100 mg/d; child: 200 mg/d)
EF = exposure frequency (365 d/yr)
ED = exposure duration (adult: 24 yr; child: 6 yr)

4.3.1.2.2 Air Inhalation. In this risk assessment, the air pathway involves the potential inhalation of fugitive dust and the inhalation of volatiles. Screening is performed for both the inhalation of fugitive dust and the inhalation of volatiles, as appropriate, for a contaminant. If a contaminant may be released to the air through fugitive dust and volatilization, the lesser (more conservative) of the two risk-based benchmark concentrations is used to evaluate contaminants. That is, the maximum detected contaminant concentration is compared to the risk-based benchmark concentrations calculated for the fugitive dust pathway and the volatile pathway. If the maximum detected concentration exceeds the smallest risk-based benchmark concentration, the contaminant is retained for further evaluation in the risk assessment. The assumptions and equations used to calculate risk-based benchmark concentrations for the inhalation pathway are provided below for the inhalation of fugitive dust and the inhalation of volatiles.

Inhalation of Fugitive Dust. For purposes of screening, several assumptions are made to determine a particulate emission factor that relates the contaminant concentration in soil with the concentration of respirable particles in the air due to fugitive dust from the contaminated site. For screening, a respirable factor (historically referred to as a particulate emission factor), which relates the concentration of a contaminant in soil to the concentration of contaminant in air, of 2E+07 m^3/kg is used in the screening equations. This factor has been derived by conservatively assuming the concentration of airborne particulates is at the maximum annual air quality standard of 0.050 mg/m^3 (40 CFR 50). It is also assumed that the concentration of contaminant in the air corresponds to the maximum contaminant concentration observed in the upper 4.6 m (15 ft) of soil and that all air particulates are derived entirely from the parent soil.

The calculation of the risk-based benchmark concentrations for the inhalation of fugitive dust are presented below. For carcinogens, the intake assumptions are a body weight of 70 kg (154 lb), an inhalation intake rate of 20 m^3/d, an exposure duration of 30 years, and an exposure frequency of 365 d/yr. The risk-based benchmark concentration in soil for noncarcinogenic effects by the fugitive dust inhalation pathway is calculated using exposure parameters for a child with an intake rate of 10 m^3/d, an exposure frequency of 365 d/yr, and an exposure duration of 6 years, as recommended in the HSBRAM (DOE-RL 1993b).

For carcinogenic non-radioactive contaminants, the screening equation is:

$$C = \frac{TR \times BW \times AT \times RF^*}{SF \times IR \times EF \times ED} \qquad 4\text{-}7$$

where:

C = risk-based benchmark concentration (mg/kg)
TR = target excess individual lifetime cancer risk (1E-07)
BW = body weight (70 kg)

AT = averaging time (365 d/yr x 70 yr)
RF = respirable factor (2E+07 m^3/kg)
SF = chemical-specific slope factor $(mg/kg\text{-}d)^{-1}$
IR = intake rate (20 m^3/d)
EF = exposure frequency (365 d/yr)
ED = exposure duration (30 yr)

*For volatiles screening, the volatilization factor (VF) is substituted for RF, as discussed below.

For noncarcinogenic effects, the equation is:

$$C = \frac{THQ \times RfD \times BW \times AT \times RF^*}{IR \times EF \times ED} \qquad 4\text{-}8$$

where:

C = risk-based benchmark concentration (mg/kg)
THQ = target hazard quotient (0.1)
RfD = contaminant-specific chronic reference dose (mg/kg-d)
BW = body weight (16 kg)
AT = averaging time (365 d/yr x 6 yr)
RF = respirable factor (2E+07 m^3/kg)
IR = intake rate (10 m^3/d)
EF = exposure frequency (365 d/yr)
ED = exposure duration (6 yr)

*For volatiles screening, VF is substituted for RF, as discussed below.

For inhalation of radioactive contaminants, via fugitive dust, the general equation to calculate risk-based benchmark concentrations is:

$$C = \frac{TR \times RF \times CF}{SF \times IR \times EF \times ED} \qquad 4\text{-}9$$

where:

C = risk-based benchmark concentration (pCi/g)
TR = target excess individual lifetime cancer risk (1E-07)
RF = respirable factor (2E+07 m^3/kg)
CF = conversion factor (1E-03 kg/g)
SF = radionuclide-specific slope factor $[(pCi)^{-1}]$
IR = intake rate (20 m^3/d)
EF = exposure frequency (365 d/yr)
ED = exposure duration (30 yr)

Inhalation of Volatiles. For volatile organic compounds with inhalation toxicity factors (Rfd's or SF's), a soil to air volatilization factor (VF) is used to define the relationship between the concentration of contaminant in soil and the amount of volatilized contaminant in air. For screening purposes, a compound was considered volatile and a VF calculated if the vapor pressure is $>10^{-6}$ mm Hg, the molecular weight is <200 and the Henry's constant is $>10^{-4}$ atm-m^3/mol. An exception to this criteria was 2-butanone (methyl ethyl ketone) with a Henry's Constant of 2.7 E-05 atm-m^3/mol, however since the Henry's constant is a measure of volatilization from water and not soil, it was carried through the screening process. These chemical specific data are presented in Table 4-9 for all volatile compounds detected within the 200-BP-1 operable unit soil (Section 4.2).

To develop the VF, it is assumed that the surface of the contaminated material is exposed directly to the atmosphere and on-site ambient air concentrations of contaminants are based directly on the emission rate of the volatile to the air from contaminated soil. The derivation of the VF is based on a model proposed in *Estimation of Multimedia Exposures Related to Hazardous Waste Facilities* (Hwang and Falco 1986). Because the VF is chemical-specific, the equation and parameters (without chemical-specific default values) are presented below for the inhalation of fugitive dust with VF substituted for RF.

$$VF\ (m^3/kg) = \frac{LS \times V \times MH}{A} \times \frac{(3.14 \times \alpha \times T)^{1/2}}{(2 \times D_{ei} \times E \times K_{as} \times CF)} \qquad \textbf{4-10}$$

$$\alpha\ (cm^2/s) = \frac{D_{ei} \times E}{E + (p_s)(1 - E)/K_{as}} \qquad \textbf{4-11}$$

where:

LS	=	width of contaminated area (m)
V	=	site-specific wind speed in mixing zone (3.3 m/s)
MH	=	mixing height (2 m)
T	=	exposure interval (s)
A	=	area of contamination (cm^2)
D_{ei}	=	effective diffusivity ($D_i \times E^{.33}$) (cm^2/s)
E	=	soil porosity (0.25 unitless)
K_{as}	=	soil/air partition coefficient (H/K_d) x 41 (g soil/cm^3 air)
CF	=	conversion factor (0.001 kg/g).
p_s	=	soil particulate density (g/cm^3)
OC	=	organic carbon content of soil (fraction)
D_i	=	chemical specific molecular diffusivity at 30°C (cm^2/s)
H	=	chemical specific Henry's Law Constant (atm-m^3/mol)
K_d	=	soil-water partition coefficient ($K_{oc} \times OC$) (cm^3/g)
K_{oc}	=	organic-carbon partition coefficient (cm^3/g).

Site specific data for V (3.3 m/s), were calculated as the annual hourly average for winds measured from the ground surface to a height of ten meters. These data were

obtained from the 200 East Area Meteorological Station. Values for E and p_s are 0.25 and 2.7 g/cm^3, respectively, and were obtained from grain size analysis data for 200-BP-1 operable unit surface soils (0 to 4.6 m [0-15 ft]). The value of OC (0.01) is from a study done on Rattlesnake Ridge within the Hanford Site (Bolton et al. 1990). Chemical specific values used to calculate the VF are provided in Table 4-9. Once the soil-to-air VF is calculated using site- and chemical-specific data, it can be used, as indicated, in the equations presented above for the fugitive dust pathway with VF substituted for RF in the equations.

4.3.1.2.3 External Exposure. The external exposure pathway applies exclusively to radioactive contaminants, which are evaluated only for their carcinogenic potential. Screening risk-based benchmark concentrations in soil are based on an exposure time of 24 hr/d, an exposure frequency of 365 d/yr, and an exposure duration of 30 yr. Certain radionuclides (e.g., Sr-90) do not emit penetrating gamma or x-rays that are the source of external exposures, and do not need to be considered for the external exposure pathway. A conversion factor of 1.14E-04 yr/hr is used, as appropriate.

Although pure beta emitters with high decay energies (e.g., strontium-90) can create penetrating photons through bremsstrahlung, the contribution of this source of radiation is considered unimportant. This is because of the presence of significant concentrations of photon-emitting radionuclides (e.g., cobalt-60 and cesium-137), or because, for pure beta emitters, the risk associated with internal exposure will be much more important than external exposure.

For radioactive contaminants, the general equation to calculate risk-based benchmark concentrations is:

$$C = \frac{TR}{SF \times IR \times EF \times ED} \qquad \textbf{4-12}$$

where:

C	=	risk-based benchmark concentration (pCi/g)
TR	=	target excess individual lifetime cancer risk (1E-07)
SF	=	radionuclide-specific slope factor [(pCi-yr/g)$^{-1}$]
IR	=	intake rate (24 hr/d x 1.14E-04 yr/hr = 2.7E-03 yr/d)
EF	=	exposure frequency (365 d/yr)
ED	=	exposure duration (30 yr)

4.3.1.3 Soils - Contaminants of Potential Concern

All contaminants identified in Section 4.2.1.2 have been carried through the risk-based screening process. Table 4-8 lists the contaminants carried through this process for both near surface soils and subsurface infiltration gravels/soils. Aluminum, magnesium, potassium, sodium, phosphate and bismuth have been eliminated from further evaluation as discussed in Section 4.3.1.1. As discussed in Section 4.3.1, the maximum value detected for each of these contaminants is compared to the risk-based benchmark screening concentration. The maximum value indicates only whether a contaminant is of potential concern, and at this point in the screening process, is not indicative of contaminant significance, distribution or extent. The results of the risk-based screening and the resulting

contaminants of potential concern are presented below for both near surface and subsurface gravels/soils.

4.3.1.3.1 Near Surface Soils (0-4.6 m [0-15 ft]). For the near surface soils, 23 of the contaminants listed in Table 4-8 are eliminated from further evaluation in the risk assessment because they are present at concentrations less than the risk-based benchmark screening concentrations. These contaminants are cadmium, copper, cyanide (total and free), nickel, selenium, thallium, nitrate, carbon disulfide, methylene chloride, benzoic acid, bis(2-ethyhexyl)phthalate, butybenzylphthalate, 2-Chloronapthalene, di-n-butylphthalate, pentachlorophenol, phenol, pyrene, 4,4'-DDT, plutonium-238, plutonium-239, and technetium-99.

Lead is eliminated from further consideration because the maximum detected concentration in surface soil (27.7 mg/kg) does not exceed the MTCA-recommended cleanup level range (i.e., 250 mg/kg) as discussed in section 4.3.1.1.

The remaining contaminants that are considered contaminants of potential concern for near surface soils at the 200-BP-1 operable unit are indicated by shading in Tables 4-10 and 4-11, and are retained for further evaluation in the baseline risk assessment. These contaminants are:

Inorganic Contaminants
- chromium

Organic Contaminants
- chrysene
- PCB

Radionuclides
- cesium-137
- manganese-54
- potassium-40
- radium-226
- strontium-90
- thorium-228
- total-U

4.3.1.3.2 Subsurface Infiltration Gravels/Soils (4.6 m [15+ ft]). For the subsurface infiltration gravels and soils, 24 of the contaminants listed in Table 4-8 are eliminated from further evaluation in the risk assessment because they are present at concentrations less than the risk-based benchmark screening concentrations. These contaminants are copper, cyanide (free), thallium, nitrate, nitrite, selenium, acetone, 2-hexanone, 4-methyl-2-pentanone, styrene, toluene, 1,1,1-trichloroethane, benzoic acid, bis(2-ethyhexyl)phthalate, butylbenzylphthalate, 2-chlorophenol, diethylphthalate, di-n-butylphthalate, di-n-octylphthalate, isophorone, pentachlorophenol, 4-4'DDT, endosulfan II, and tritium.

Lead is eliminated from further consideration because the maximum detected concentration in subsurface soil (24.3 mg/kg) does not exceed the MTCA-recommended cleanup level (i.e., 250 mg/kg) as discussed in Section 4.3.1.1.

The remaining contaminants that are considered contaminants of potential concern for subsurface infiltration gravels/soils at the 200-BP-1 operable unit are indicated by

shading in Tables 4-12 and 4-13, and are retained for further evaluation in the baseline risk assessment. These contaminants are:

Inorganic Contaminants
- cadmium
- chromium
- cyanide (complex)
- cyanide (total)
- nickel

Organic Contaminants
- tributyl phosphate
- PCB
- aldrin

Radionuclides
- antimony-125
- cesium-137
- chromium-51
- cobalt-60
- plutonium-238
- plutonium-239
- plutonium-239+240
- potassium-40
- radium-226
- strontium-90
- technetium-99
- thorium-228
- total U

4.3.2 Groundwater - Risk-based Screening Methodology

4.3.2.1 Risk-based Screening Approach and Assumptions

All contaminants identified in Section 4.2.2.2 have been carried through the risk-based benchmark screening process. These contaminants are listed in Table 4-14. In order to give qualitative perspective on significance, Table 4-14 also lists the maximum concentration detected for each contaminant, the total number of detects and the total number of samples analyzed.

For the groundwater pathway, groundwater ingestion is the only exposure scenario evaluated. This is appropriate as the inhalation pathway is not applicable and the duration of exposure is so short that external exposure via radionuclides is negligible. The exposure assumptions are discussed in Section 4.3.2.2 and summarized in Table 4-7.

Aluminum does not have an EPA-endorsed toxicity value and is generally considered harmful only in relation to medical therapy (Amdur et al. 1991). Therefore, aluminum is eliminated from further evaluation. Calcium, magnesium, iron, potassium and sodium are essential nutrients and are therefore also eliminated from further evaluation (EPA 1989a; EPA-10 1991 and DOE-RL 1993b). Bismuth and silicon are also eliminated because bismuth is generally not considered toxic and silicon is non-toxic and ubiquitous in the environment (Amdur et al., 1991).

Turbulent flow conditions necessary for particulate transport in groundwater are artificially induced within an aquifer during well purging and sampling. As a result suspended but settleable particulates (turbidity) observed in groundwater samples are not representative of actual aquifer conditions, and reflect human disturbance of the system while obtaining a sample. EPA guidance (EPA 1986a) suggests an upper limit on groundwater turbidity of 5 NTU for representative samples when analysis is sensitive to turbidity. Field analysis for turbidity indicated many samples had turbidity levels in excess of 5 NTU and variations were also observed within sampled groundwater from a given well among sampling events. Turbidity is important for the inorganic parameters because metals are contained within the suspended solids. Filtered samples ensure more representative samples when turbidity levels are unacceptably high. Filtering a water sample ensures consistent turbidity and, therefore comparability among samples within a particular well and among wells in the same aquifer. For the 200-BP-1 operable unit groundwater sampling two samples for each well, representing filtered (0.45 μm filter) and unfiltered samples, were obtained for analysis of TAL metals. For the risk-based screening process, only filtered samples are used for the inorganic parameters (metals). For the radionuclides, non-filtered samples were used because out of approximately 215 samples, only four were filtered. Turbidity values measured during sampling are included in Appendix E-3 (Field Parameters).

Additionally, radionuclide testing included analysis of gross alpha and gross beta concentrations within groundwater samples. These measurements indicate the presence and overall magnitude of radionuclide concentrations. However, toxicity information for radionuclides is isotope specific, therefore measurements of gross alpha and gross beta cannot be evaluated in this risk assessment. Numerous radionuclides are present in the soil in the 200 BP-1 operable unit, all of which would contribute varying levels of alpha and beta radiation. Radionuclides anticipated to be present from operations in the site were analyzed. Daughter products can be estimated based on the concentration of the parent.

4.3.2.2 Risk-based Screening Calculations

The risk-based screening calculations are presented below for the groundwater ingestion pathway. Example calculations are provided in Appendix I.

4.3.2.2.1 Groundwater Ingestion. All water ingestion risk-based benchmark concentrations are derived using residential exposure assumptions from the HSBRAM (DOE-RL 1993). For carcinogens, the exposure assumptions are based on adult exposure. The adult exposure factors are a body weight of 70 kg, a daily intake rate of 2 L/d, an exposure frequency of 365 d/yr, and an exposure duration of 30 years. For water ingestion of contaminants with noncarcinogenic effects, the exposure assumptions are based only on exposures for a child. The exposure factors, therefore, are a body weight of 16 kg, a water ingestion contract rate of 1 L/d, and exposure frequency of 365 d/yr, and an exposure duration of 6 yr.

For carcinogenic non-radioactive contaminants, the screening equation is:

$$C = \frac{TR \times BW \times AT}{SF \times IR \times EF \times ED} \qquad 4\text{-}13$$

where:

C = risk-based concentration (mg/L)
TR = target excess individual lifetime cancer risk (1E-07)
BW = body weight (70 kg)
AT = averaging time (365 d/yr x 70 yr)
SF = chemical-specific slope factor $(mg/g\text{-}d)^{-1}$
IR = intake rate (2 L/d)
EF = exposure frequency (365 d/yr)
ED = exposure duration (30 yr)

For non-carcinogenic, the screening equation is:

$$C = \frac{THQ \times RFD \times BW \times AT}{IR \times EF \times ED} \qquad 4\text{-}14$$

where:

C = risk-based benchmark concentration (mg/L)
THQ = target hazard quotient (0.1)
RfD = contaminant-specific chronic reference dose (mg/kg-d)
BW = body weight (16 kg)
AT = averaging time (365 d/yr x 6 yr)
IR = intake rate (1 L/d)
EF = exposure frequency (365 d/yr)
ED = exposure duration (6 yr)

4.3.2.2.2 Groundwater - Contaminants of Potential Concern All contaminants identified in Section 4.2.2.2 have been carried through the risk-based screening process and are listed in Table 4-14. Aluminum, calcium, magnesium, potassium, sodium, silicon and bismuth have been eliminated from further evaluation as discussed in Section 4.3.3.1. The results of the risk-based screening for the 200 BP-1 operable unit groundwater and the resulting contaminants of potential concern are summarized in Table 4-15.

For groundwater, 13 contaminants are not considered contaminants of potential concern and are eliminated from further consideration because they are present at concentrations less than the risk-based benchmark screening concentrations. These contaminants are mercury, strontium, tin, ammonia, chloride, 2-butanone, 1-3 dichlorobenzene, 2-chlorophenol, benzoic acid, phenol, butybenyphthalate, fluoranthene, and pyrene.

Lead, chloride, and sulfate do not have toxicity values (RfDs or SFs) with which to calculate a risk-based screening concentration. Therefore, the national primary and secondary Drinking Water Regulation Standards (maximum contaminant levels [MCLs]) are used to indicate whether lead, chloride or sulfate are a potential contaminant of concern. In general the HSBRAM risk-based benchmark screening concentrations are much more conservative than the MCLs. MCLs are used for screening only when risk-based benchmark screening concentrations cannot be calculated.

The remaining contaminants that are considered contaminants of potential concern for 200-BP-1 operable unit groundwater are indicated by shading in Table 4-15. These contaminants are:

Inorganic Contaminants

- antimony
- arsenic
- barium
- beryllium
- cadmium
- cyanide (total)
- cyanide (free)
- cyanide (complex)
- chromium
- copper
- lead
- manganese
- nickel
- selenium
- silver
- thallium
- vanadium
- fluoride
- nitrate
- nitrite
- sulfate

Organic Contaminants

- trichloroethene
- 4,4'-DDT

Radionuclides

- cobalt-60
- potassium-40
- plutonium-238
- radium-226/228
- strontium-90
- technetium-99
- total uranium
- tritium

4.4 EXTENT OF CONTAMINATION

The intent of this section is to look at the spatial distribution of the contaminants of potential concern identified in Section 4.3 for both soils and groundwater at the 200-BP-1 operable unit. For soil, a qualitative analysis of contaminant frequency, distribution and chemical activity will eliminate a few contaminants of potential concern from further evaluation in the risk assessment (Chapter 6). This step is a final, qualitative screening process, to focus the list of contaminants of potential concern on the contaminants which drive the risk for the 200-BP-1 operable unit. The 95% upper confidence limits (UCLs) of the mean soil concentrations are presented at the end of this section.

For groundwater, plume maps are presented to indicate the locations and extent of major contaminant plumes. Historical trends in contaminant concentration are discussed in terms of significant contaminant trends and temporal changes. Groundwater

contaminants of potential concern will not be addressed in this report beyond this section, therefore, UCLs are not calculated for the identified groundwater contaminants of potential concern. These groundwater contaminants will be addressed in the 200 East Aggregate Area Groundwater study.

4.4.1 Soils

Sampling of 200 BP-1 operable unit soils took place during two different sampling events. The first sampling event was part of Task 3 (Surface Soil Sampling). Sampling consisted of ground surface samples collected throughout the 200-BP-1 operable unit (Figure 2-3). Following this sampling event the surface soils were scraped and piled on top of the cribs and then capped with "clean" soil.

The second sampling suite was part of Tasks 2 and 4 of the Phase I RI. These events consisted of the collection of grab samples from boreholes drilled through the cribs to depths ranging from 9 to 15 m (29.5 to 50 feet) (3 boreholes were drilled to depths of about 70 m [230 ft]). All drilling for Tasks 2 and 4 was conducted on the cribs and penetrated through the cover soils placed on the cribs during the interim stabilization activity. Additionally, spectral gamma ray logging was also performed in selected Task 2 and 4 boreholes. The Task 2, 3, and 4 sampling and the gamma-ray logging results are discussed below for the contaminants of potential concern identified in Section 4.3.1 for both the near surface soil and the subsurface infiltration gravels/soils.

4.4.1.1 Near Surface Soils (0-4.6m [0-15 ft])

Inorganics

Chromium was detectable at a concentration above the risk based screening level for inhalation. The maximum detected chromium concentration in near-surface soils is below the Hanford Site-wide soil background concentration of 27.9 mg/kg. Therefore, chromium was eliminated as a contaminant of potential concern because it is indistinguishable from background in the near-surface soils.

Organics

For the near surface soils, the only organic compounds detected that were above risk based screening levels are chrysene and PCBs. Both of these organic analytes were only detected in one sample at 0.04 mg/kg and 0.022 mg/kg, respectively. Since these organic analytes were only detected once and at concentrations near the detection limits, they have been eliminated as a contaminant of potential concern.

Radionuclides

The type of radionuclide contamination detected with 200-BP-1 operable unit near surface soils is similar within the nine cribs. Table 4-16 shows the maximum contaminant concentrations within the three borehole (BH) samples, for each respective crib. It also lists the total number of detects for each contaminant within all nine cribs (i.e. for all nine cribs combined). Cesium-137, potassium-40, radium-226, strontium-90 and thorium-228 were detected in each of the cribs with at least 87 total detects for each contaminant (Table 4-16). Total uranium was detected in cribs 216-B-44, 216-B-46, 216-B-47, 216-B-48, 216-B-50 and 216-

B-57 with a total of 57. All the contaminants are distributed fairly evenly with respect to depth, with no obvious horizons of increased radionuclide contamination.

The radionuclide contamination detected within the soils sampled during Task 3 varies from that detected in the Task 2 and 4 sampling events. Maximum concentrations for cesium-137, potassium-40, radium-226, thorium-228, and total uranium are higher than those found in the Task 2 and 4 sampling events (Table 4-16). However, the highest levels of cesium-137, radium-226 and total uranium found within the Task 2 and 4 sampling events are found in the 0.6-1.8 m (2-6 ft) sample zone. Therefore, it seems likely that this contamination may be from soils placed on top of the cribs during Task 3.

Also within the Task 3 sampling, manganese-54 was also found at concentrations exceeding its risk-based benchmark screening concentration for the external exposure pathway. Manganese-54 was detected only once, and has a half life of less than one year. However, the detection of manganese-54 with a short half-life, in the top few inches of soil, may indicate decay products of recent fallout from other Hanford sources. Due to the frequency of detection and extremely short half life manganese-54 will not be considered further in the risk assessment (Chapter 6).

Potassium-40 had only one detect, at 18.7 pCi/g within the Task 3 soils, which exceeds area-specific background concentration (UTL) of 18.5 pCi/g. Since there was only one detect, which is very close to background, and because potassium-40 is a naturally occurring radionuclide, it will not be considered further in the risk assessment (Chapter 6).

4.4.1.2 Subsurface Infiltration Gravels/Soils (>4.6 m [>15 ft])

Inorganics

Cadmium, chromium, cyanide (total) and nickel are the only four inorganic contaminants which exceeded their respective risk-based benchmark screening concentrations. For chromium only two detects were above the background concentrations. The maximum detect (119 mg/kg at crib 216-B-43 (BH 299-E33-296) at a depth range of 68-69.5 m [223-228 ft]) was close to the water table and may have resulted from groundwater contamination, possibly from other operable units. The other sample with detected chromium above the background concentration of 27.9 mg/kg occurred at 9 to 10 m (30 to 32 ft) at a concentration of 29.5 mg/kg. Since chromium was slightly above background in one crib soil sample, chromium will not be considered a contaminant that could pose a risk from 200-BP-1 operable unit sources.

Total and complex cyanide was found to exceed the risk-based benchmark screening concentration of 160 mg/kg in only one sample (248.5 mg/kg at crib 216-B-47 (BH 299-E33-320) at a depth of 7-7.9 m [23-26 ft]). All other detects were well below the risk-based benchmark screening concentration. All toxicity values with which to evaluate cyanide are for free cyanide, which was analyzed for in the sampling suite. Therefore, total and complex cyanide exceed screening for only one sample and only if the oral Rfd for free cyanide is used as a surrogate. This approach is unnecessarily conservative since free cyanide was analyzed for in the same depth interval at crib 47 and was found at 1.9 mg/kg, well below the risk-based screening concentration. Therefore, total and complex cyanide will not be considered further in the risk assessment.

Organics

For the subsurface gravels/soils, three contaminants, PCB, tributyl phosphate and aldrin had maximum detected contaminant concentrations which exceeded their respective risk-based bench mark screening concentrations. PCBs were detected twice in crib 216-B-50 (BHs 299-E33-308 and 299-E33-303) at 4.9-5.5m (16-18 ft) and once in cribs 216-B-49 (BH 299-E33-302) and -57 (BH 299-E33-304) at 5.2-6.1 m (17-20 ft) and 4.6-5.2 m (15-17 ft), respectively. Tributyl phosphate was tentatively identified at least 28 times in cribs 216-B-43 through -50 and in all but two cases was found in samples between 4.6-10.7 m (15-35 ft). Only one detect (93 ug/kg at crib 216-B-43 (BH 299-E33-314) at a depth of 5.5-6.4 m [18-21 ft]) was above the risk-based benchmark screening concentration of 40 mg/kg. Aldrin was detected only once in crib 216-B-49 at 5.2-6.1 m (17-20 ft). Detected concentration of 6.6. ug/kg is below contract required detection limit (CRQL) of 8 ug/kg, therefore, aldrin will not be considered further in the risk assessment.

Radionuclides

The radionuclide contamination concentrations detected within the 200-BP-1 operable unit subsurface infiltration gravels/soils (>4.6 m [>15 ft.]), are much higher than those detected in the surface soils. Table 4-17 shows the maximum contaminant concentrations within the three boreholes sampled, for each respective crib. It also lists the total number of detects for each contaminant within all nine cribs (i.e., for all nine cribs combined). Antimony-125, cobalt-60, plutonium 238, 239, 239+240 and technetium-99 which were not detected in the near surface soils, are found consistently in the sub-surface infiltration gravels and underlying native soils. The specific contaminants detected in each of the cribs are somewhat variable, only plutonium 238 and 239, strontium-90, and technetium-99 are found consistently in each of the cribs (Table 4-17).

Crib 57 has significantly lower concentrations of strontium-90 and cesium-137. This is consistent with historical records that show cribs 216-B-57 and 216-B-50 received evaporative condensate as opposed to actual liquid waste. However, the contamination in crib 216-B-50 is consistent with cribs 216-B-43-49 and may, in fact, have received supernatant decant liquid waste from the BY Tank Farm.

Two unexpected radionuclides, chromium-51 and antimony-125, with half lives of 28 days and 2.1 years, respectively, were detected in subsurface soils at concentrations exceeding the risk-based benchmark screening concentration for the external exposure pathway. Chromium-51 was detected once each at cribs 216-B-45 (BH 299-E33-319) and 216-B-49 (BH 299-E33-313) at a depth of 8.2-9.1 m (27-30 ft). Additionally, chromium-51 is not a daughter product that could be generated as decay from other radionuclide species. Therefore, due to frequency of detection, the short half life and the fact that the external pathway is the only one of concern for contaminants which are located at depth, chromium-51 will not be considered further in the risk assessment. Antimony-125, however, was detected a total of 10 times in several different cribs at depths between 7.6-18 m (25-59 ft). Antimony-125 is also not a daughter product generated from decay of other major species; however it will be carried through the risk assessment due to the frequency of detection.

Potassium-40 is eliminated from further consideration below 4.6 m (15 ft) because the maximum concentration detected (16.9 pCi/g) is below the area-specific background value (UTL) of 18.5 pCi/g.

The largest detected concentrations of radionuclides are between (4.6-9.1 m [15-30 ft.]) below the ground surface, although contamination was detected up to 71.9 m (236 ft.) below the surface. For strontium-90, cesium-137, plutonium-238 and 239+240, and total uranium maximum concentrations are consistently between 4.9-7.9 m (16-26 ft) and mostly between 5.4-6.7 m (18-22 ft) as shown in Table 4-17. The concentration of these radionuclides at the bottom of the cribs is consistent with their relative immobility (high partitioning with soil relative to water phases) and may also indicate that these radionuclides were discharged to the cribs as a suspended flocculent within the supernatant effluent.

The data and the contrasts between the near surface and subsurface gravels/soils are consistent with the crib construction and waste disposal methods utilized. Based on information from the sample drilling operation, the top and bottom of the cribs (engineered infiltration gravels) are generally from 3.4-6.1 m (11-20 ft) below ground surface, respectively. The waste discharge pipes were designed to discharge effluent below the top of the crib gravels and infiltrate through approximately 1.8 m (6 ft) of gravels before reaching presumably in-situ soil at the crib bottom. Samples were generally taken at the top and bottom soil/gravel and gravel/soil interface areas, respectively, for each crib. All the samples taken in the soil/gravel interface at the top of the cribs (typically 2.7-4 m [9-13 ft]) show much less contamination than samples taken at the gravel/soil interface at the bottom of the crib. The few samples taken at 4.6 m (15 ft) (the cutoff between near surface and subsurface zones for this report), contain significantly lower levels of radionuclides than samples in the same crib taken a few feet deeper. This is illustrated in Table 4-18 which shows contaminant levels by depth for two cribs with both near surface and subsurface samples close to 4.6 m (15 ft). Gross alpha and gross beta are included in Table 4-18 as overall indicators of radionuclide activity.

4.4.1.3 Spectral Gamma-Ray Logging. Geophysical logging was performed on 27 boreholes in the 200-BP-1 operable unit using a Radioactive Logging System (RLS). The RLS consists of a high resolution, high purity germanium (HPGe) passive spectral gamma-ray system. This was the first significant field test of the RLS. Three types of boreholes were logged, including newly constructed groundwater monitoring wells, existing groundwater monitoring wells undergoing remediation, and the vadose zone boreholes drilled through the cribs. A description of the activity and the results of surveys are presented in Price (1992). The boreholes which were logged using this technique are summarized in Table 2-4.

The objective of the borehole surveys was to characterize the presence and distribution of man-made gamma-emitting radionuclides in soils beneath the crib area using a geophysical technique. Gamma emissions for four radionuclide species were monitored, including cesium-137, cobalt-60, antimony-125, and europium-154. Total gamma, which was also measured, is the count rate total, in counts per second (cps), for all the detected gamma-rays. The results presented in Price (1992) include graphs depicting the decay activities (concentrations) versus depth for each borehole. Decay activities are presented in picocuries per gram (pCi/g) of soil.

As reported in Price (1992), there is uncertainty associated with the results of the spectral-gamma ray surveys. Causes of this uncertainty include the following:

- In order to convert the gamma energy reading into radioisotope concentrations, the spectral gamma-ray system must be calibrated. Calibration

requires the use of a number of simplifying assumptions regarding source homogeneity and the borehole environment, and the use of correction factors to compensate for certain non-ideal borehole conditions, such as the presence of water and the use of steel casing. No correction factors were available, however, to compensate for the presence of grout between multiple casing strings, formation seals, casings of different materials, and cumulative casing thicknesses greater than 1 cm (0.40 in.).

- In high activity zones, the counter frequently became saturated. To address this problem, the tool was modified to incorporate the use of a lead shield. This improved the system capabilities on the upper end, however, use of the shield in high activity zones limited the systems' ability to detect radioelements at low decay activities.

- Multiple borehole surveys in vadose zone hole 299-E33-304 indicated that drilling operations may have significant effects on the radionuclide profiles measured by the RLS. Radionuclide movement in the casing annular spaces was suggested as the possible cause of significantly different activity profiles in the same borehole as drilling progressed.

In addition, the extent of gamma penetration through soils and its effect on the measured radionuclide distributions is not clearly understood. In the vicinity of the highly radioactive intervals, gamma penetration may lead to the apparent detection of decay activity for some distance above and below the actual zone of contamination. The distance to which this occurs is uncertain, but may be anywhere from approximately 0.3-1.5 m (1-5 ft). For this reason, as well as those listed above, the concentration profiles and depths of occurrence presented in this report must be regarded as semi-quantitative indicators of specific gamma-emitting radionuclides and their distributions.

RLS data plots for boreholes within cribs 216-B-43 through -50 were generally similar. As an example, plots from two cribs, 216-B-43 (borehole 299-E33-296) and 216-B-46 (borehole 299-E33- 311) are presented in Figures 4-1 and 4-2, respectively. Of the 27 surveys completed, these two are considered representative of the deep and shallow RLS survey results. Review of the two figures suggests the following general comments with respect to the extent of radionuclide contamination beneath the crib area:

- Soils above approximately 3.7-4.6 m (12-15 ft) are characterized by relatively low radionuclide levels, as compared to the crib gravels and deeper zones. Total gamma levels above 3.7-4.6 m (12-15 ft) were generally less than 100 cps.

- The levels increase sharply at a depth of approximately 3.7-4.6 m (12-15 ft), and in most cases go off-scale for total gamma (greater than 1,000 cps) and Cs-137 (5000 pCi/g). Levels remain off-scale until a depth of about 9-15 m (30-50 ft) and then continue to decline until a depth of approximately 30 m (100 ft). Below 30 m (100 ft) levels remain uniformly low.

- Of the four radionuclides measured, Cs-137 is present to a much greater extent than the others. It accounts for essentially all of the total gamma activity detected above 15 m (50 ft), where total gamma levels are highest. Below 30 m (100 ft), however, Cs-137 was generally not detected.

- Co-60 was generally present, but at much lower levels and greater depths than Cs-137. Concentrations were generally below 10 pCi/g. In crib 216-B-43, Co-60 was detected from approximately 40-58 m (130-190 ft). The maximum level in the crib was approximately 4 pCi/g. The low levels observed and greater depths of occurrence probably are reflective of the radionuclide's greater mobility in the subsurface environment (Section 4.4.2).

- Antimony-125 and europium-154 were detected infrequently, in isolated zones and at relatively low levels, as compared to Cs-137. Among the crib boreholes, europium-154 was detected only in crib 216-B-44 (borehole 299-E33-297). An exception is at crib 216-B-49 (borehole 299-E33-302) where antimony-125 was detected from a depth of approximately 6-27 m (20-90 ft) and at concentrations that reached nearly 50 pCi/g.

In addition to the surveys which were performed in the Task 2 and 4 Coreholes, existing monitoring wells were logged as well (Table 2-4). The results of these surveys differ from the survey results obtained in the crib boreholes in that significant levels of contamination were detected at depth (>30 m [100 ft]). A low level of confidence is associated with these results, however, because the competency of the surface seals in these wells is doubtful. Contamination may have migrated to depth due to inadequate seals. A higher degree of confidence must be associated with the new wells because of the strict procedures which were followed during drilling to prevent vertical leakage through annular spaces.

The radionuclide distribution observed in crib 216-B-57 is depicted in Figure 4-3. This crib is addressed separately from cribs 216-B-43 through -50 because of its greater size, physical location away from the other cribs, and waste effluent source (ITS condensate). Significant contaminant characteristics indicated in the figure include the following:

- Cs-137 was the only radionuclide detected in the crib of the four which were measured.

- Significant gamma emissions were not detected in the borehole until a depth of approximately 10 m (30 ft). This is considerably deeper than in the case of the 216-B-43 through -50 cribs where radiation levels generally rose sharply at a depth of approximately 3 m (10 ft).

- As with the 216-B-43 through -50 cribs, after reaching maximum values at depths of around 9 - 15 m (30 - 50 ft), contaminant levels declined until a depth of approximately 30 m (100 ft). Beneath this depth levels remained uniformly low. None of the radionuclides were detected below this depth.

- As reported in Price (1992), no man-made radionuclides were detected in 299-E33-306 (borehole C in crib 216-B-57). This borehole is located at the opposite end (north end) of crib 216-B-57, as shown in Figure 2-1. As the crib is a long, linear trench, which slopes to the north, these results indicate that the infiltration of waste effluent occurred primarily in the southern portion. This is the end of the crib where the influent piping was located. Apparently waste flow to the north end was limited or did not occur due to infiltration in the upstream portion of the crib.

Despite the uncertainties mentioned earlier for the RLS, the radionuclide distributions which were measured by the spectral-gamma ray logs are generally consistent with the analytical results obtained from the soil sampling activities. While the concentrations and depths of occurrence measured by the RLS must be considered as general indicators only, the magnitudes of the radionuclides measured and the depth intervals where they were observed by the RLS compare well with the soil analytical data. Tables 4-16 and 4-17 summarize the maximum detected radionuclide concentrations obtained from the surface and subsurface soil sampling activities, respectively. Table 4-18 indicates selected radionuclide levels with depth. As seen in the tables, radioisotope levels remained low (below about 15 pCi/g) until a depth of approximately 4.6 m (15 ft) where the concentrations rose dramatically. Maximum concentrations for most radionuclides occurred at depths between 4.6 and 9.1 m (15 and 30 ft). Co-60 levels were generally below 5 pCi/g and maximum concentrations occurred at greater depths than for Cs-137.

One of the data objectives of the 200-BP-1 Phase I RI work plan was an evaluation of the possible occurrence of perched groundwater conditions during crib operations. Perching could have occurred if there existed a relatively continuous and thick layer of fine-grained material beneath the cribs. Such a layer, if it was sufficiently impermeable, could impede the vertical movement of waste downward through the unsaturated zone, potentially leading to a perched water table. This was of concern because the occurrence of perching could indicate that contaminants moved laterally beyond the immediate crib area. The adequacy of the current groundwater monitoring network at the 200-BP-1 operable unit would need to be re-evaluated if reasonable evidence existed that such migration may have occurred. The results of the spectral gamma-ray logging are useful to determine whether the subsurface contaminant distribution is indicative of groundwater perching during crib operation.

In order to assess whether or not perching occurred during crib operation, the results of the spectral gamma-ray logging from three wells are examined. These wells include 299-E33-13, 299-E33-38 and 299-E33-40. The locations of these wells are indicated in Figures 2-4 and 3-1. These wells constitute RLS surveyed boreholes which are not located immediately within the crib area. Since it is expected that waste infiltrating into the cribs would have moved primarily vertically (unless perching were to induce lateral flow), detection of radioisotopes in these three boreholes could indicate lateral flow, especially if the contaminants were associated with a suspected fine-grained layer.

The graphs depicting the radioisotope concentrations measured in the RLS surveys are presented in Price (1992). The results indicate the following:

- For well 299-E33-13, the measured gamma decay activities are uniformly low for the entire depth of the borehole. There is no obvious zone of contamination which would indicate that waste effluents migrated laterally to the vicinity of the well. The well is approximately 90 m (300 ft) southeast from the nearest crib (216-B-43).

- For well 299-E33-38, two areas of increased decay activity were observed in the survey. These consist of zones at depths of approximately 16.8 m (55 ft) and 57.6 m (189 ft). These areas are characterized by localized increases in decay activity (spikes) that occur over an approximately 0.3-0.9 m (1-3 ft) zone. The zone at 16.8 m (55 ft) does not appear to be associated with a fine-grained layer, based on an examination of the boring log (Hoffman

1992). The zone at 57.6 m (189 ft), however, does occur at approximately the same depth as a fine-grained layer which apparently is continuous beneath the cribs, as shown in Figure 3-23. This zone at a depth of 58m (190ft) was discussed earlier in Chapter 3 as the only continuous fine-grained layer observed beneath the cribs. The well is located approximately 30 m (100 ft) south from the nearest crib (216-B-43).

- For well 299-E33-40, there are no distinct zones of contamination. Contamination is detected at relatively low, but uniform levels from a depth of approximately 30 m (95 ft) until a depth of 43 m (140 ft). No fine-grained intervals were noted in the boring log over this depth range. The presence of the contamination at 299-E33-40 does not, therefore, appear to be related to lateral migration due to perching. The contamination could have resulted from the slight lateral spreading (dispersion and diffusion) which likely occurred during infiltration. This well is located approximately 15 m (50 ft) from the recent crib (216-B-50).

The results of the spectral gamma-ray logging indicate that perching does not appear to have occurred to a significant extent beneath the 200-BP-1 operable unit. While some evidence was obtained that perching may have occurred at a depth of approximately 58 m (190 ft), the extent of this lateral migration appears to have been limited to the operable unit boundaries. The generally coarse-grained soils and lack of major fine-grained intervals has encouraged primarily vertical waste infiltration without significant lateral spreading in the vadose zone.

4.4.1.4 Soil Contaminants of Potential Concern. Table 4-19 indicates all of the contaminants of potential concern identified in Section 4.3, and the reasons why they were eliminated in Section 4.4. Table 4-20 lists the contaminants of potential concern for both near surface soil and subsurface infiltration gravels/soils that will be carried forward and evaluated in Chapter 6. Based on the fact that crib 216-B-57 received waste as evaporative condensate and has significantly lower concentrations of strontium-90 and cesium-137 than cribs 216-B-43 to -50, it will be treated separately in the remaining section of this risk assessment. While records indicate crib 216-B-50 received only evaporative condensate, the contamination is similar to cribs 216-B-43 to -49 which received decanted supernatant effluents. Therefore it will be grouped with cribs 216-B-43 to -49. Table 4-20 also contains the 95% upper confidence limits (UCLs) of the mean soil concentrations for each contaminant of potential concern. For the near surface soils, UCLs were calculated using the samples taken between 0-4.6 m (0-14.9 ft). For the subsurface, the first two samples taken below 4.6 m (15 ft) (usually 4.6-10.7 m [15-35 ft]) were utilized in UCL calculation. Average concentrations (UCLs) of subsurface soils between 4.6 and 10.7 m (15 and 35 ft) are needed to estimate risk to receptors from pathways due to surface disruption in the risk assessment (Chapter 6).

4.4.2 Groundwater

This section characterizes the current and historic nature and extent of groundwater contamination throughout the study area. The current extent of groundwater contamination is depicted in a series of plume maps shown in Appendix K (Figures K-1 through K-11). Maps have been prepared for selected contaminants of potential concern identified in Section 4.3.

In addition, historic contaminant data are compiled and presented below for a number of key study area wells. Selected indicator contaminants are examined at the wells to qualitatively assess trends and the temporal variation in groundwater contamination. Although the groundwater contaminants of potential concern are evaluated herein for their spatial and temporal distributions, the list, as presented in Table 4-15, will not be modified further.

4.4.2.1 Historical Data Trends at Selected Key Wells. Historic contaminant data, as well as the recent Phase I RI data, have been compiled for selected constituents and at selected wells in order to assess the prior contaminant level variations which have occurred throughout the study area. Data are considered for the following wells:

- 299-E33-7
- 699-49-55a
- 699-49-57a
- 699-50-53a
- 699-53-55a
- 299-E33-12

These wells were chosen because, as a group, they provide a relatively broad areal coverage across the study area, have a long sampling history, and include both unconfined and confined aquifer wells. Well 299-E33-7 is located directly within the immediate crib area and is assumed to be generally representative of groundwater beneath the cribs. Well 699-53-55a is located near the downgradient boundary of the study area. Wells 699-49-55a, 699-49-57a and 699-50-53a are located near the middle of the study area, at an approximately intermediate distance between 299-E33-7 and 699-53-55a. These wells are located in the middle portions of the current contaminant plumes which are emanating from the 200 East Area, as is discussed in Section 4.4.2.2.

Well 299-E33-12 is constructed within the confined aquifer and is located immediately east of the 200-BP-1 operable unit. Until the early 1980s, due to poor well construction, the well is thought to have acted as a flow conduit which permitted direct hydraulic communication between the unconfined and confined aquifers (Connelly et al. 1992). Waste effluents present in the unconfined aquifer in the vicinity of the cribs are believed to have migrated through well 299-E33-12 into the confined aquifer. The mechanism for this migration likely consisted of the movement of a denser than water, high-salt waste effluent through the annular space between the basalt and the well casing (Smith 1980; Graham et al. 1984). Groundwater mounding beneath operational cribs may have created local downward hydraulic gradients that may have also contributed to the contaminant migration from the unconfined to the confined aquifer.

The result of this interaquifer communication which is associated with well 299-E33-12 is that the well acted as a point of contaminant introduction into the Rattlesnake Ridge confined aquifer. Contaminant levels measured in the well during the period of crib operation, as discussed below, closely mirrored levels observed in unconfined aquifer wells directly beneath the cribs. Trends observed at the well are therefore important because they provide indications of the extent to which the well acted as a contaminant source to the Rattlesnake Ridge confined aquifer.

For the selected wells discussed above, trend plots have been prepared for the following contaminants:

- gross beta
- cesium-137
- selenium
- strontium-90

- chloride
- chromium
- cobalt-60
- total cyanide
- nitrate
- sulfate
- technetium-99
- tritium
- uranium

These constituents were chosen because they include contaminants of potential concern for groundwater in the study area, and because they include other significant indicator parameters (e.g., chloride, sulfate) which are helpful in the identification of contaminant plume movement.

The trend plots are depicted in Figures 4-4 through 4-16. The data that is depicted was obtained from the Hanford Site groundwater database and represents the period from 1955 throughout the first five quarterly sampling events conducted under the Phase I RI. The quantity of data varies among constituents. No analyses are available for several contaminants until the late 1980s, such as chromium, cyanide, selenium, and Tc-99. Chloride and sulfate results were not reported until the mid-1970s. And for several constituents, considerable gaps are present where no data were collected for periods of years. For gross beta, Co-60, nitrate, and tritium, however, a relatively complete and continuous record is available.

It should be emphasized that much of the data presented was collected some years ago and for a variety of purposes, was based on varying analytical methods and procedures, and have not been subjected to the level of quality assurance control and validation which is required of current data collection efforts. The accuracy and precision of much of the data is, therefore, uncertain. For the purposes of this qualitative assessment, however, particularly since contaminant trend variations of interest are typically orders of magnitude, use of the data is considered appropriate. If historical trends are important during the risk assessment for the 200 East Aggregate Area Groundwater Study, it is suggested that a thorough review of the quality of the historic data be conducted. This review was also recommended by the 200 Aggregate Area Management Study and should include both sampling procedures and analytical methods.

Based on a review of the contaminant plots presented in Figures 4-4 through 4-16 the following major trends and significant data features have been identified. This information is based on a qualitative assessment of the data and presents general data trends observed:

- As expected, contaminant levels in source area groundwater, as represented by concentrations at well 299-E33-7, were greatly elevated during the period of crib operation. Gross beta concentrations were typically on the order of 10^6 to 10^7 pCi/L. Co-60 concentrations were in the range of 10^5 to 10^6 pCi/L. Nitrate, Cs-137 and Sr-90 as well, were quite high. Other contaminant levels were also likely elevated; the data, however, is limited with respect to constituent levels during the period of crib operation. No information is available during this time period (for well 299-E33-7), for example, for tritium, Tc-99, chromium, selenium, and cyanide, among others.

- While levels still remained significantly elevated above background at the end of the period examined, constituent concentrations at the source area (well 299-E33-7) had dropped dramatically, generally by several orders of

magnitude or more. Gross beta was reduced to the 10^2 to 10^3 pCi/L range. Co-60 was lowered to 10^1 to 10^2 pCi/L.

- During the period when the cribs were operating, especially during the late 50s and early 60s, contaminant levels at confined aquifer well 299-E33-12 were similar in magnitude to the levels measured in the unconfined aquifer. As seen with gross beta, Cs-137, Co-60, nitrate, Sr-90 and uranium, and probably for other compounds as well, the concentration levels and variations at well 299-E33-12 closely reflected those observed at well 299-E33-7. This indicates that the well served to introduce contaminated groundwater from beneath the crib area directly to the Rattlesnake Ridge confined aquifer. Currently, levels in well 299-E33-12 are generally low with respect to the other wells examined, except for Tc-99 and Co-60 which are higher than within the unconfined aquifer.

- At the end of the period evaluated, the highest contaminant levels of the five wells considered were generally at well 699-50-53a, followed by wells 699-49-55a and 299-E33-7. Generally the lowest levels were observed at the downgradient well 699-53-55a and in the Rattlesnake Ridge aquifer well 299-E33-12. Despite being some 3.5 km (2.2 mi) downgradient of the 200 East Area, contaminants have migrated to well 699-53-55a, albeit at greatly reduced levels compared to other wells. The contaminant plumes extend at least to 699-53-55a.

- Contaminant levels at well 699-50-53a were highly elevated as early as 1959 when the well was apparently first sampled. High levels of gross beta, Cs-137, Co-60, nitrate, Sr-90 and tritium were observed at 699-50-53a in 1959 and 1960. In most cases these were the highest concentrations ever measured at the well. For well 699-49-55a, relatively high tritium levels were observed in 1962 when the well was apparently first sampled. By 1959, therefore, contaminant plumes had migrated at least as far as the 0.7 km (0.45 mi) which separates well 699-50-53a from the 200 East Area.

- Contaminant levels at wells 699-50-53a and 699-49-55a generally were at maximum values throughout the 1950s and 1960s and then generally declined throughout the 70s and early 1980s. Over the period 1982 to 1986, however, depending on the constituent, levels began to increase again in the two wells. This occurrence was observed for gross beta, chloride, Co-60, nitrate and sulfate. For cyanide and Tc-99, it may have occurred as well; the data, however, is not sufficient to define the trend definitively. For Cs-137 it only occurred at 699-49-55a. Generally, this trend was not observed at wells 299-E33-7 and 699-49-57a, where levels generally declined over the entire observation period. The trend was observed, however, at the two wells for tritium. The cause of this observed sudden contaminant increase is not clear but may be due to the movement of a distinct contaminant plume into the study area over the period of approximately 1982 to 1986. For well 699-49-55a, levels generally began to decrease again in 1987 suggesting that the main body of the plume had passed. At 699-50-53a, however, the levels remained generally high throughout the late 1980s and early 1990s.

- A similar occurrence was observed at well 699-49-57a for nitrate in 1973. Nitrate levels increased at the well in a step-wise fashion, rapidly increasing from about 1-2 mg/L to 200-300 mg/L. A similar increase was not observed at any other well or for any other constituent.

- In contrast to contaminant levels for a number of the compounds observed, which varied by several orders of magnitude or more between wells and over the observation period, levels of uranium did not vary greatly. While limited data was available for uranium, historic levels appeared to remain relatively constant regardless of the well. Levels were slightly elevated at 299-E33-7 and 299-E33-12 in the 1950s over later concentrations, however, there was little variation over time from well to well. All values reported were approximately within one order of magnitude.

- The levels measured during the Phase I RI data collection effort were generally consistent and continuous with the earlier data, as seen in the figures.

4.4.2.1 Contaminant Plumes. This section describes the current extent of groundwater contamination at the 200-BP-1 operable unit study area, based on data collected during the Phase I RI. Data is presented below for the contaminants of potential concern identified in Section 4.3. The discussion of the extent of contamination is organized as follows:

- Plume maps have been created for contaminants of potential concern which displayed well-defined groundwater plumes, i.e. for those compounds which were consistently detected throughout the study area. The plume maps are included in Appendix K. The following compounds were mapped:

  cyanide (total)
  cyanide (free)
  vanadium
  nitrate
  cobalt-60
  potassium-40
  technetium-99
  total uranium
  tritium
  gross alpha
  gross beta

- The remaining contaminants consist of those compounds which were detected infrequently or at only one or two wells. These compounds, because of their limited extends, do not exhibit plumes of marked areal extent. Thus, rather than creating plume maps, the extent of contamination for these compounds has been presented in tabular form (Table 4-21). The table lists all sample detects (above background and screening values) for each compound (for all wells and sampling events). As seen in the table, 22 of the 32 contaminants of potential concern fall within this category. Each compound in the table is characterized by generally fewer than 3 total detects.

4.4.2.1.1 Contaminants of Limited Extent. The contaminants of potential concern which display limited extent are presented in Table 4-21. No further discussion of these compounds will be included.

4.4.2.1.2 Plume Maps. Maps for contaminants of potential concern which are characterized by extensive plumes are presented in Appendix K. The maps are based on the analytical data collected during the Phase I RI. The values plotted for each well represent the average of all quarterly sampling concentrations. In situations where all of the values for a particular well were above the detection limit, the mean was calculated normally. Where some of the values were below detection, and some above, the average was calculated by assuming a concentration of one-half the detection limit for those samples below the detection limit. In situations where all of the samples were below the detection limit, the average was considered to be 0 and is shown on the maps as non-detected (ND). The maps are based on the 200-BP-1 project database as of November 20, 1992. Significant features depicted in these maps are discussed below.

Cyanide (total). Total cyanide is located beneath and to the north of the 200-BP-1 operable unit at concentrations ranging from 9 to 965 µg/L (Figure K-1). Concentrations above the risk-based screening value of 32 µg/L are located in two areas, both of which are north of the 200-BP-1 operable unit. One of the areas is centered approximately at well 699-50-53 where an average concentration of 964.7 µg/L was observed. This value was the maximum value observed in the study area. The shape and extent of the plume suggests cyanide migration from the 200-BP-1 operable unit. The other area is centered in the vicinity of well 699-55-57 where a total cyanide level of 91.9 µg/L was measured.

Cyanide (free). Free cyanide is localized to the vicinity of well 699-50-53 where an average concentration of 131 ppb was measured (Figure K-2). The risk-based screening value for free cyanide in this RI is 32 µg/L.

Nitrate. Nitrate is present in large, really extensive portions of the 200-BP-1 study area at concentrations above the background level of 12.4 mg/L (DOE-RL 1992b). The detected values range from 1 to 493 mg/L (Figure K-3). The US EPA primary drinking water MCL for nitrate (as nitrate) is 45 mg/L. Concentrations above the MCL of 45 mg/L are primarily beneath and northeast of the 200-BP-1 operable unit for both the confined and unconfined aquifers. The highest concentrations detected were at wells 699-50-53a and 699-49-55b with 492 and 493 µg/L, respectively. The portion of the unconfined aquifer plume that extends into the study area from the west is likely from a 200 West Area source.

Vanadium. Vanadium is apparently ubiquitous throughout the study are at concentrations ranging from 11 to 27 µg/L (Figure K-4). The risk-based screening level is 11 µg/L, and the Hanford site Groundwater Background is 15 µg/L (DOE-RL 1992b). The highest concentrations are 26.8 ppb and 27.8 ppb at wells 699-E33-38 and 699-53-55.

Gross alpha. The US EPA primary drinking water MCL for gross alpha is 15 pCi/L. Concentrations above 15 pCi/L are found north of the 200 East Area for the unconfined aquifer and to the north and east of the 200 BP-1 operable unit for the confined aquifer (Figure K-5). Detected concentrations range from 1 to 17.1 pCi/L with the maximum of value detected at well 699-49-55a. Wells 699-49-57, 699-49-55, and 699-55-57 are characterized by quarterly average levels of approximately 17 pCi/L. Contamination in the confined aquifer is associated with well 299-E33-12.

Gross beta. Widespread gross beta contamination is present beneath the 200-BP-1 study area, with detected levels ranging from 6 to 5180 pCi/L (Figure K-6). The US EPA primary drinking water MCL is 50 pCi/L. For the unconfined aquifers, two primary areas of contamination are centered at wells 699-50-53a and 699-55-57. The levels measured at these

two wells were 5180 and 972 pCi/L, respectively. For the confined aquifer, the maximum concentration was at well 699-E33-12 (952 pCi/L)

Cobalt-60. Co-60 contamination is centered at well 699-50-53a where the quarterly average value was 364 pCi/L (Figure K-7). Detected concentrations ranged from 5 to 364 pCi/L. The risk-based screening value for Co-60 in this RI is 0.3 pCi/L. Additionally, well 699-55-57 exhibited a concentration of 67 pCi/L. In the confined aquifer, a concentration 43.6 pCi/L was observed at 299-E33-12.

Potassium-40. Detected concentrations of potassium-40 ranged from 66 to 159 pCi/L. Most of the detects are located in two general areas, underneath and to the south of the 200-BP-1 operable unit, and in the northern portion of the 200-BP-1 study area (Figure K-8). The highest concentration was in well 699-E33-15 (159 pCi/L). The risk-based screening value for K-40 in this RI is 0.42 pCi/L. Potassium-40 is a naturally occurring radionuclide and is present in all groundwater. Hanford Site background values of K-40 are not currently known but may explain its generally ubiquitous extent. In the confined aquifer, a value of 125 pCi/L was observed at 299-E33-12.

Technetium-99. Tc-99 is widespread in the 200-BP-1 study area (Figure K-9). The detected concentrations range between 8 and 15000 pCi/L. The risk-based screening value for Tc-99 is 3.5 pCi/L. There are two primary areas of contamination centered at wells 699-50-53a and 699-55-57. Tc-99 levels observed at these two wells were 15000 and 1860 pCi/L, respectively. Contamination in the confined aquifer was observed at wells 299-E33-12 (1251.8 pCi/L) and 699-47-50 (204.2 pCi/L).

Total uranium. The risk-based screening value for total uranium in this RI is 0.16 pCi/L. 2.64 µg/L is the background value determined in the Hanford Site Groundwater Background (DOE-RL 1992b). Detected concentrations of total uranium range from 1 to 14 µg/L, and concentrations above background are located in plumes north and east of the 200 BP-1 operable unit (Figure K-10). The highest concentration outside of the 200-BP-1 operable unit was observed at well 699-E32-2 (14 µg/L). Contamination was observed in the confined aquifer at levels above background of wells 699-49-55b, 299-E33-40, and 699-47-50 (3, 4.3 and 3.1 µg/L, respectively).

Tritium. Tritium contamination in the 200-BP-1 study area is widespread and covers the largest area of any of the contaminants of potential concern. Levels beneath the 200-BP-1 operable unit and immediately to the north are typically in the 1000 to 7000 pCi/L range. The highest levels observed were at wells 699-E32-2 (166000 pCi/L) and 699-E33-24 (15800) (Figure K-11). The risk-based screening value (at 10^{-7} risk level) for tritium in this RI is 85 pCi/L. Tritium was detected in confined aquifer wells 299-E33-12 (734 pCi/L), and 699-47-50 (178 pCi/L).

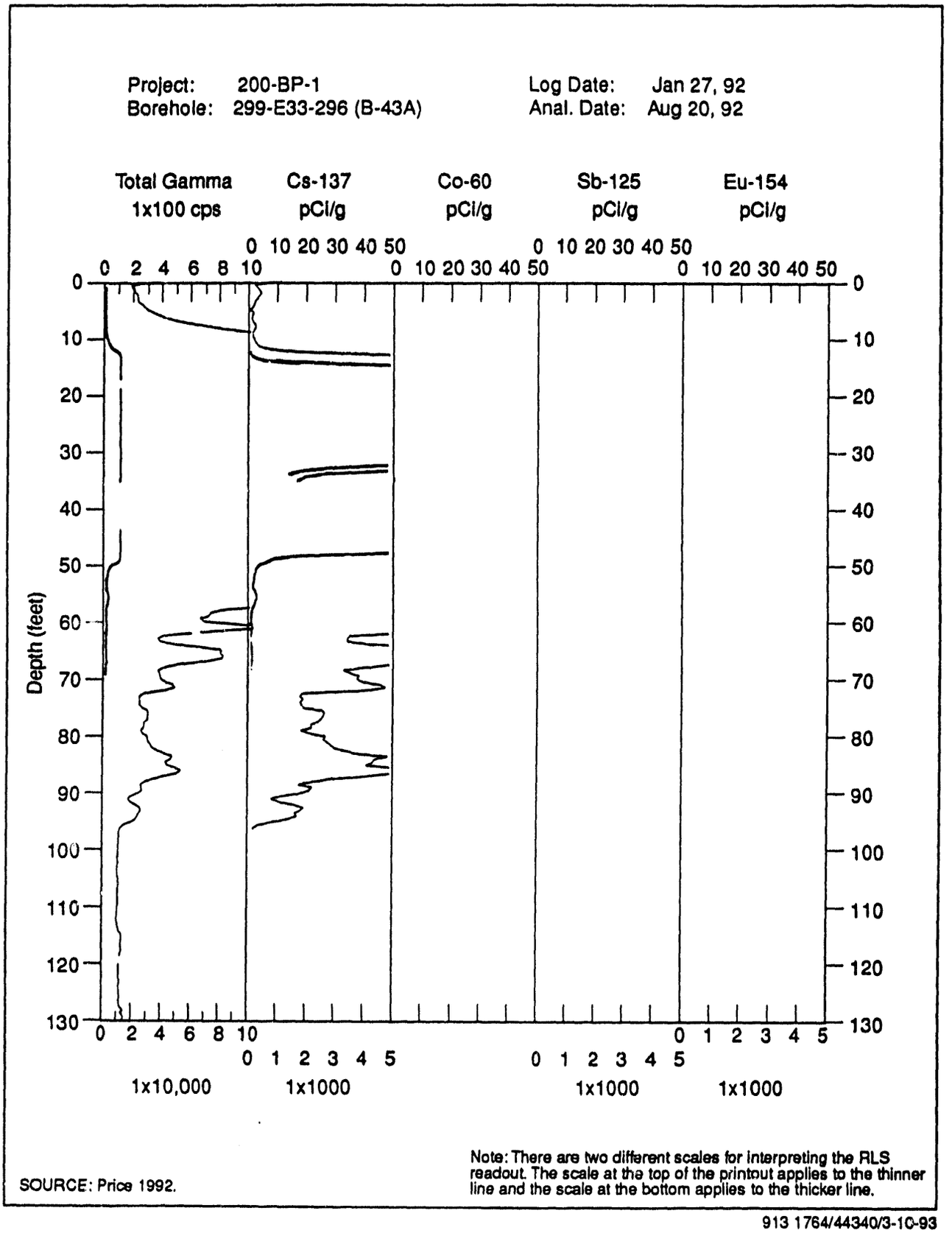


Figure 4-1a. RLS Spectral Gamma-Ray Survey of Borehole 299-E33-296 (216-B-43A). (Sheet 1 of 2)

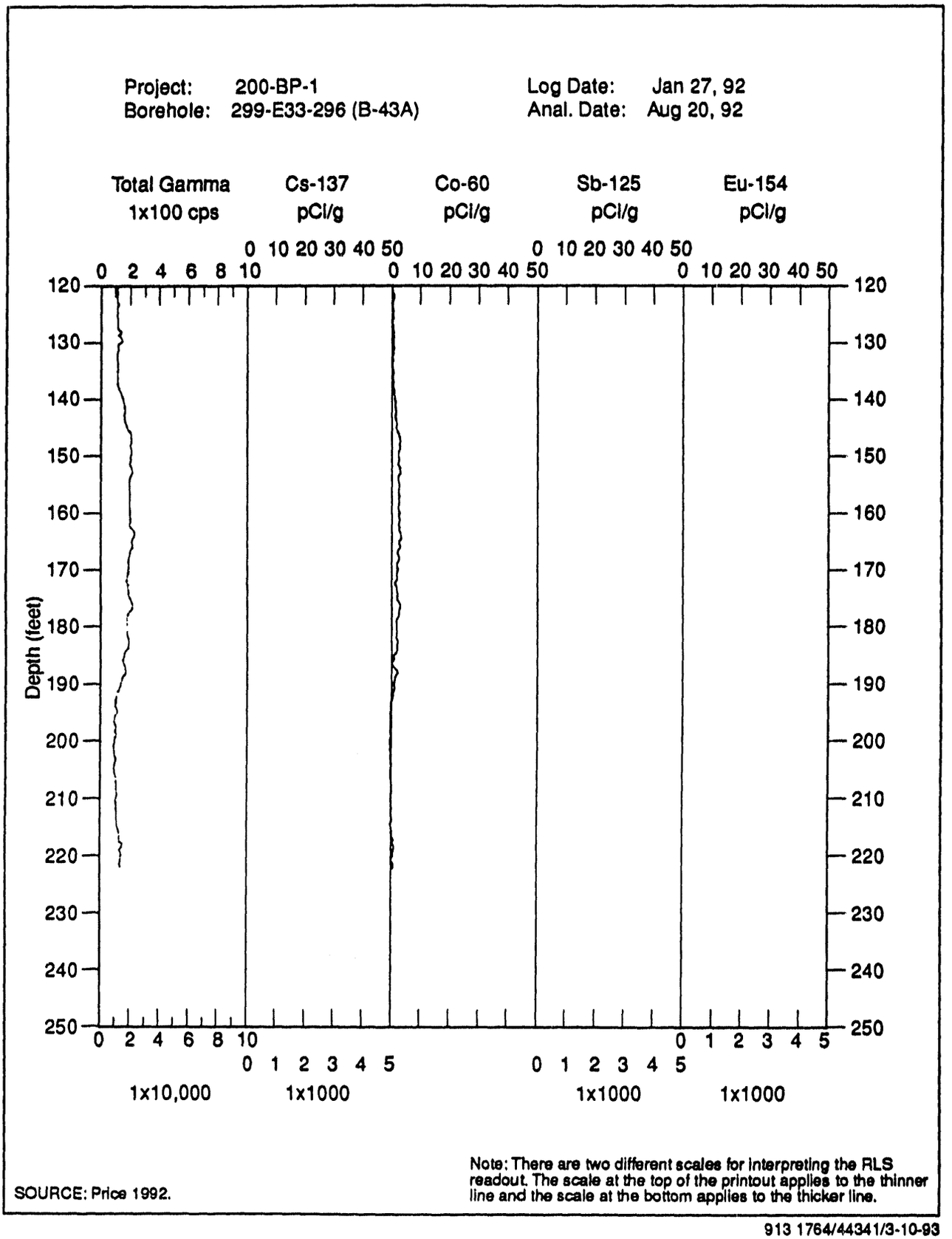
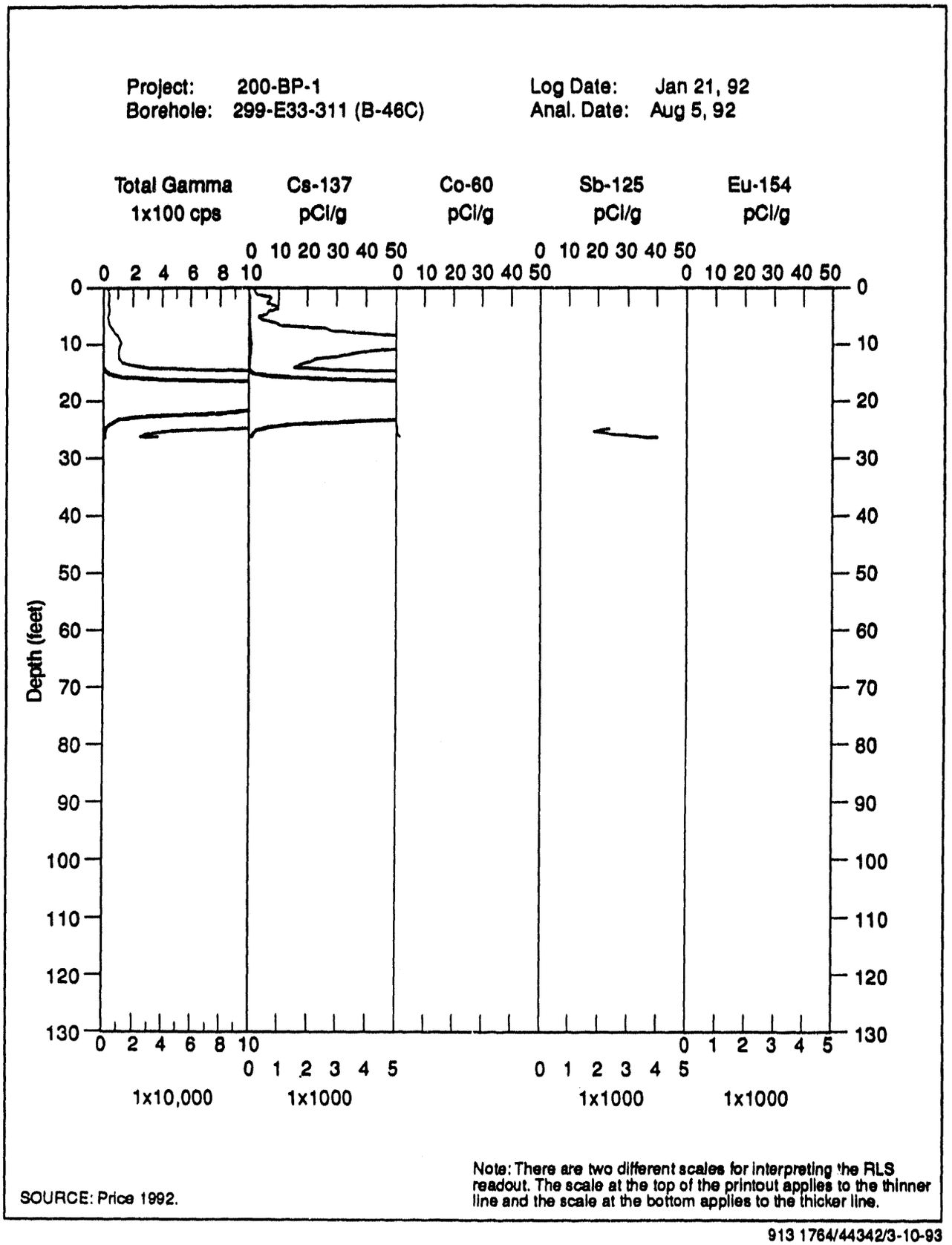


Figure 4-1b. RLS Spectral Gamma-Ray Survey of Borehole 299-E33-296 (216-B-43A). (Sheet 2 of 2)



913 1764/44342/3-10-93

Figure 4-2. RLS Spectral Gamma-Ray Survey of Borehole 299-E33-311 (216-B-46C).

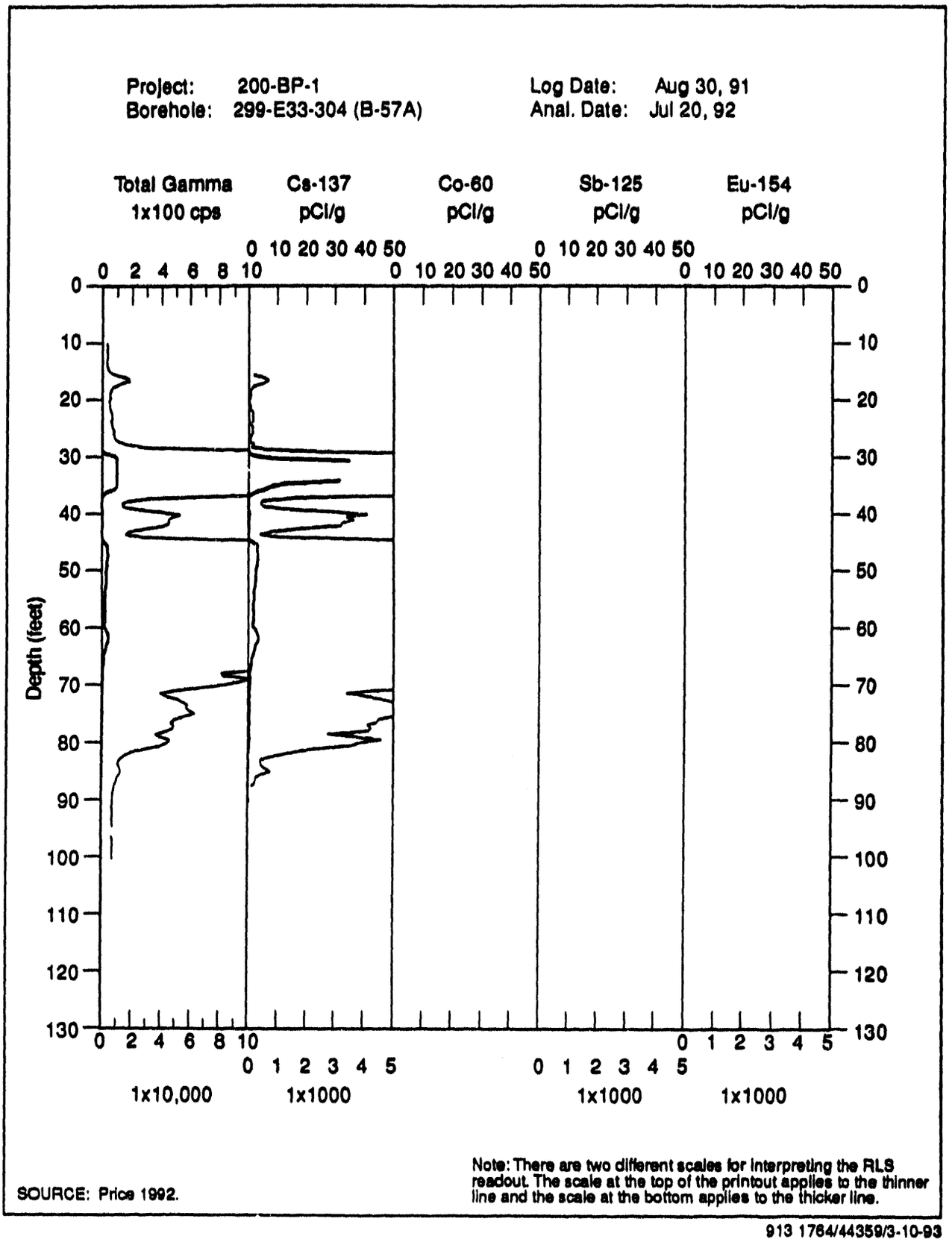


Figure 4-3a. RLS Spectral Gamma-Ray Survey of Borehole 299-E33-304 (216-B-57A). (Sheet 1 of 2)

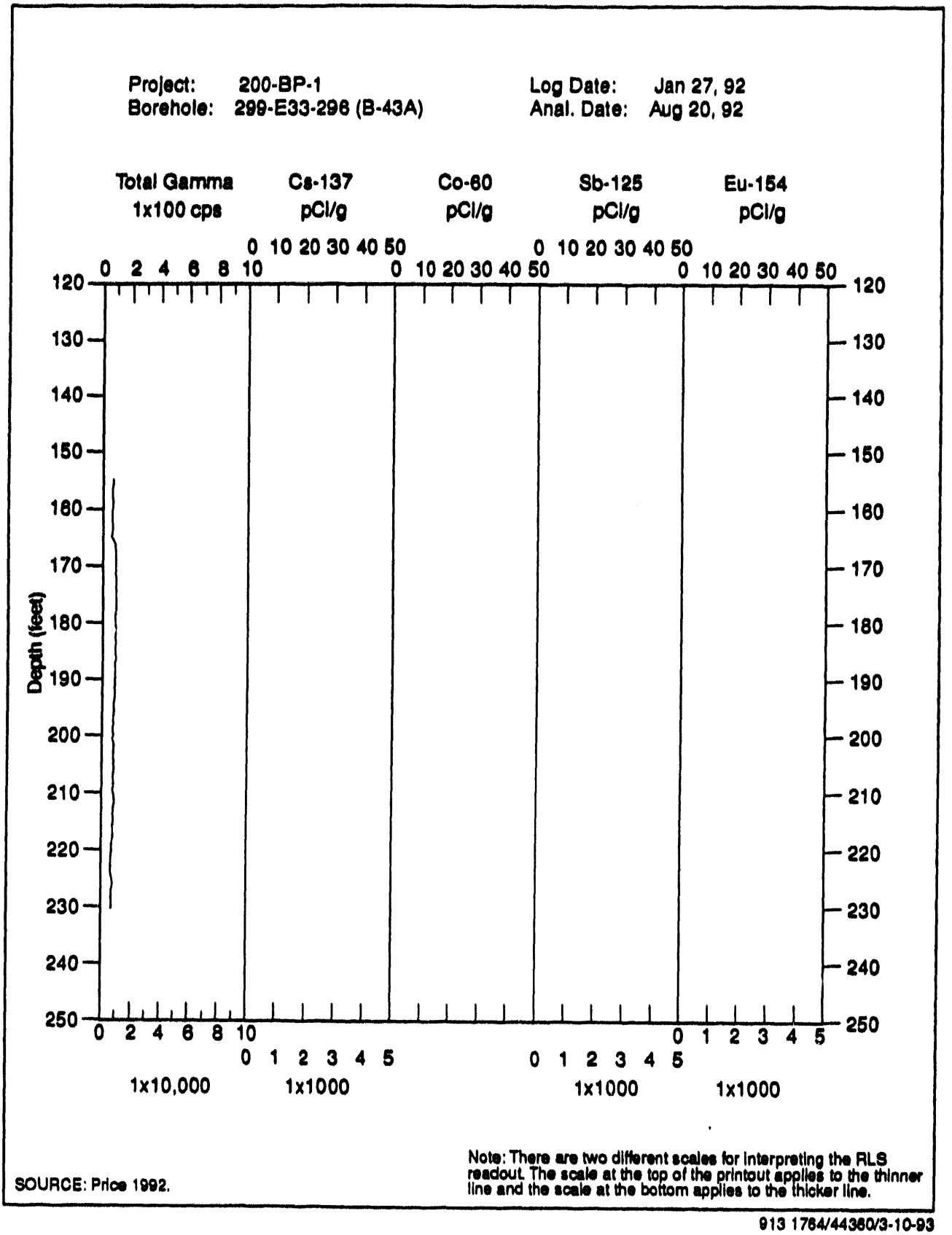
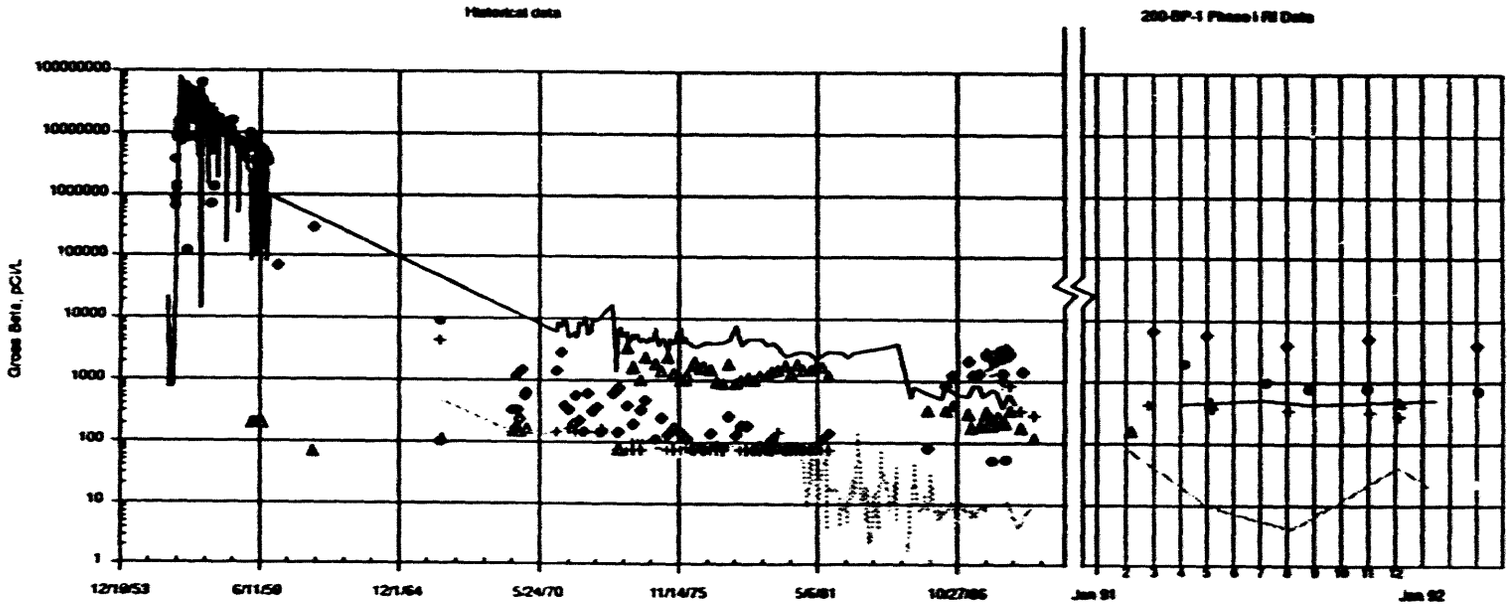


Figure 4-3b. RLS Spectral Gamma-Ray Survey of Borehole 299-E33-304 (216-B-57A). (Sheet 2 of 2)

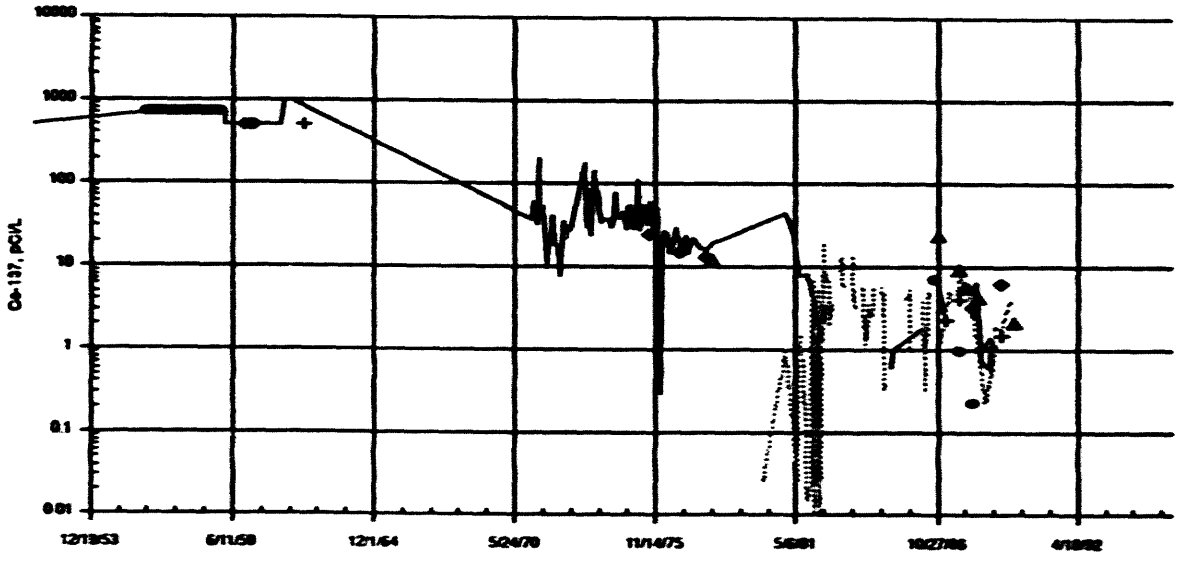


NOTE: Data plotted was obtained from the WHC groundwater database. Data is presented to indicate general temporal trends in contaminant levels. Much of the data, especially that collected prior to 1990, however, were obtained without the current level of quality assurance control. The precision and accuracy of these data is uncertain.

The use of lines for wells E33-7, 53-55a does not imply continuity of data. For example between 1960 and 1970 data were not collected.

913 1764/44252/3-10-93

Figure 4-4. Historical Gross Beta Concentrations in Selected Wells.



* All the available new data are less than the Contract Required Detection Limits (CRDL).

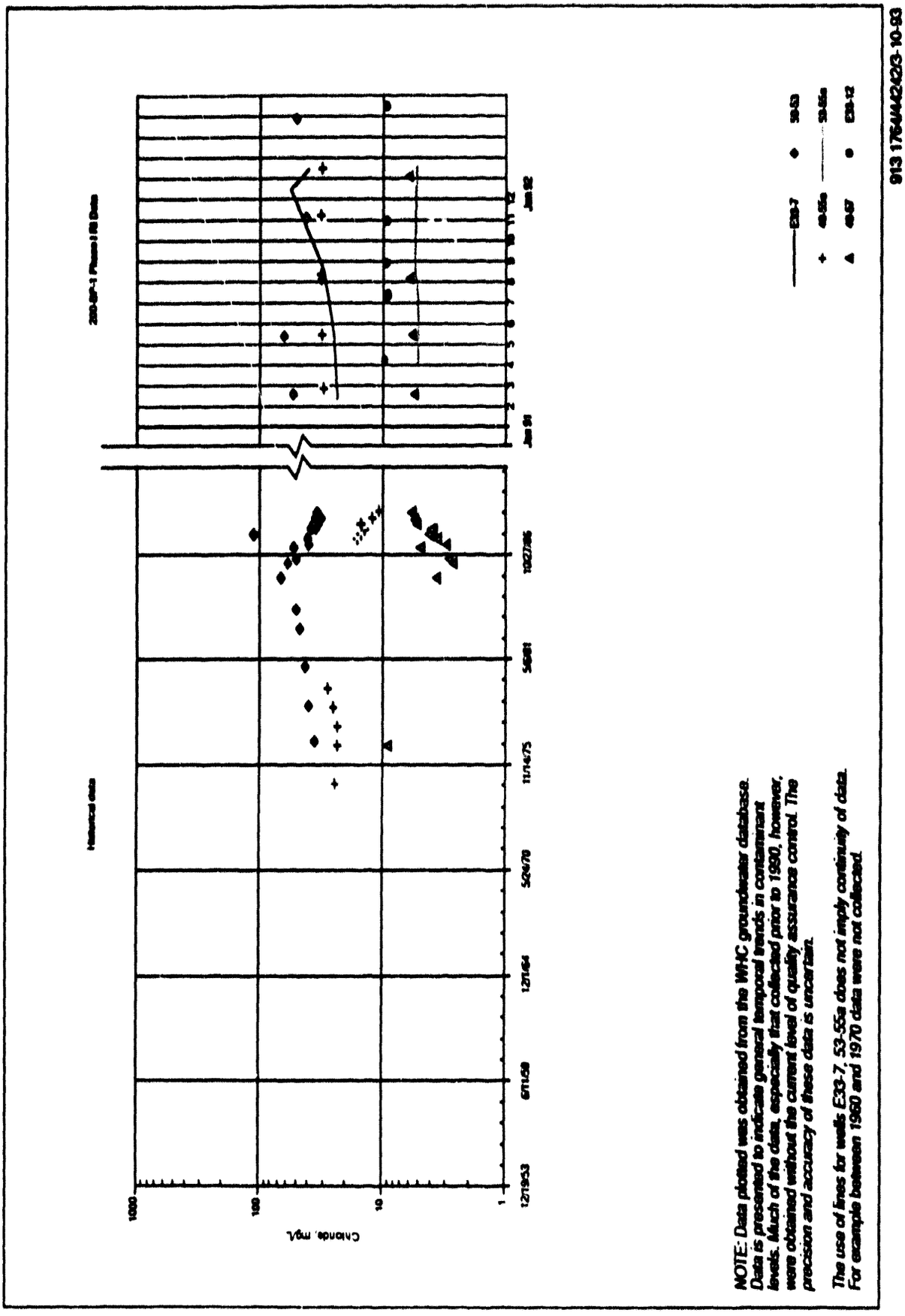
NOTE: Data plotted was obtained from the WHC groundwater database. Data is presented to indicate general temporal trends in contaminant levels. Much of the data, especially that collected prior to 1990, however, were obtained without the current level of quality assurance control. The precision and accuracy of these data is uncertain.

The use of lines for wells E33-7, 53-55a does not imply continuity of data. For example between 1960 and 1970 data were not collected.

E33-7 ◆ 58-53
\+ 49-65a 53-55a
▲ 49-67 ● E39-12

913 1764/44251/3-10-93

Figure 4-5. Historical Cs-137 Concentrations in Selected Wells.



NOTE: Data plotted was obtained from the WHC groundwater database. Data is presented to indicate general temporal trends in contaminant levels. Much of the data, especially that collected prior to 1990, however, were obtained without the current level of quality assurance control. The precision and accuracy of these data is uncertain.

The use of lines for wells E33-7, 53-55a does not imply continuity of data. For example between 1960 and 1970 data were not collected.

Figure 4-6. Historical Chloride Concentrations in Selected Wells.

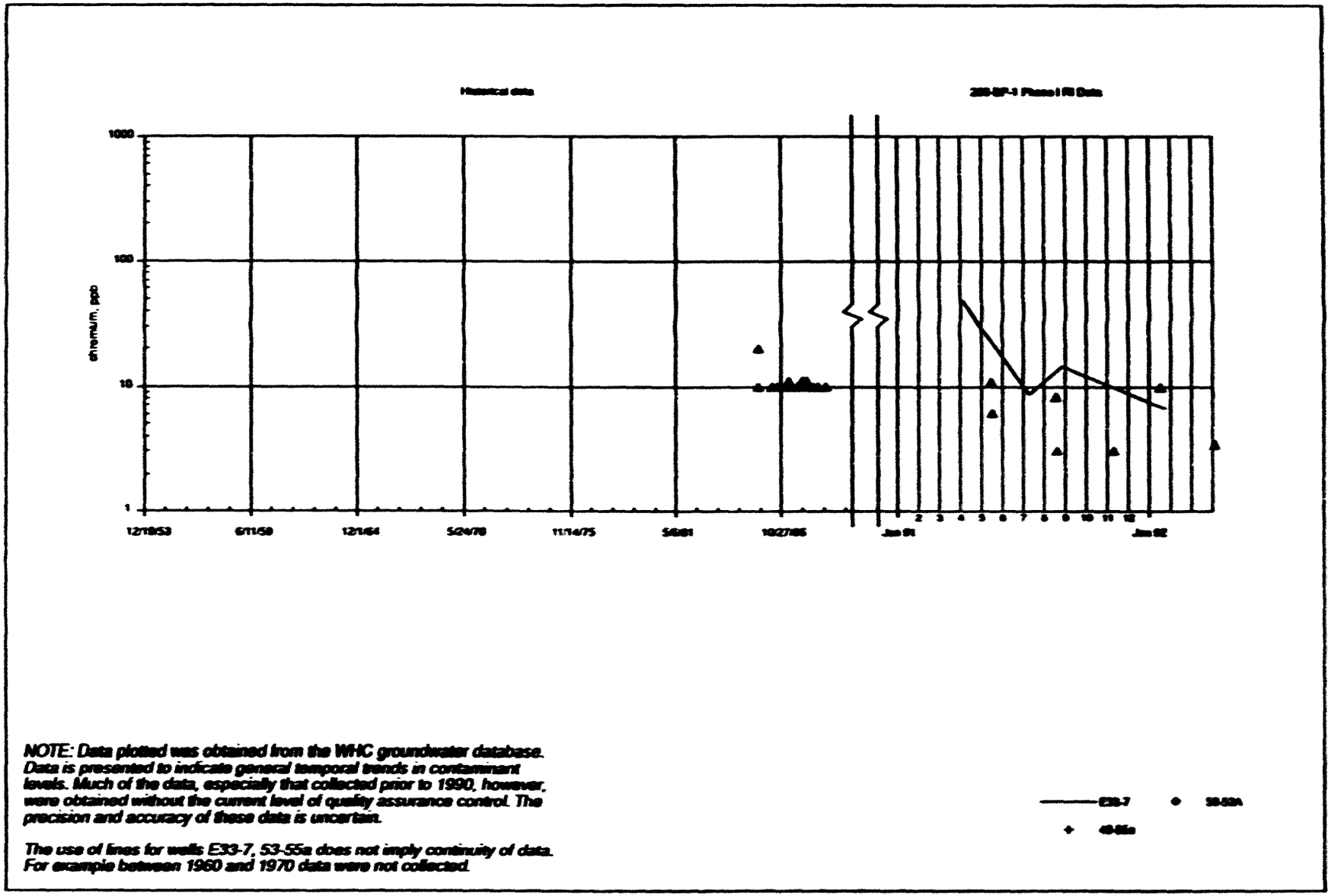
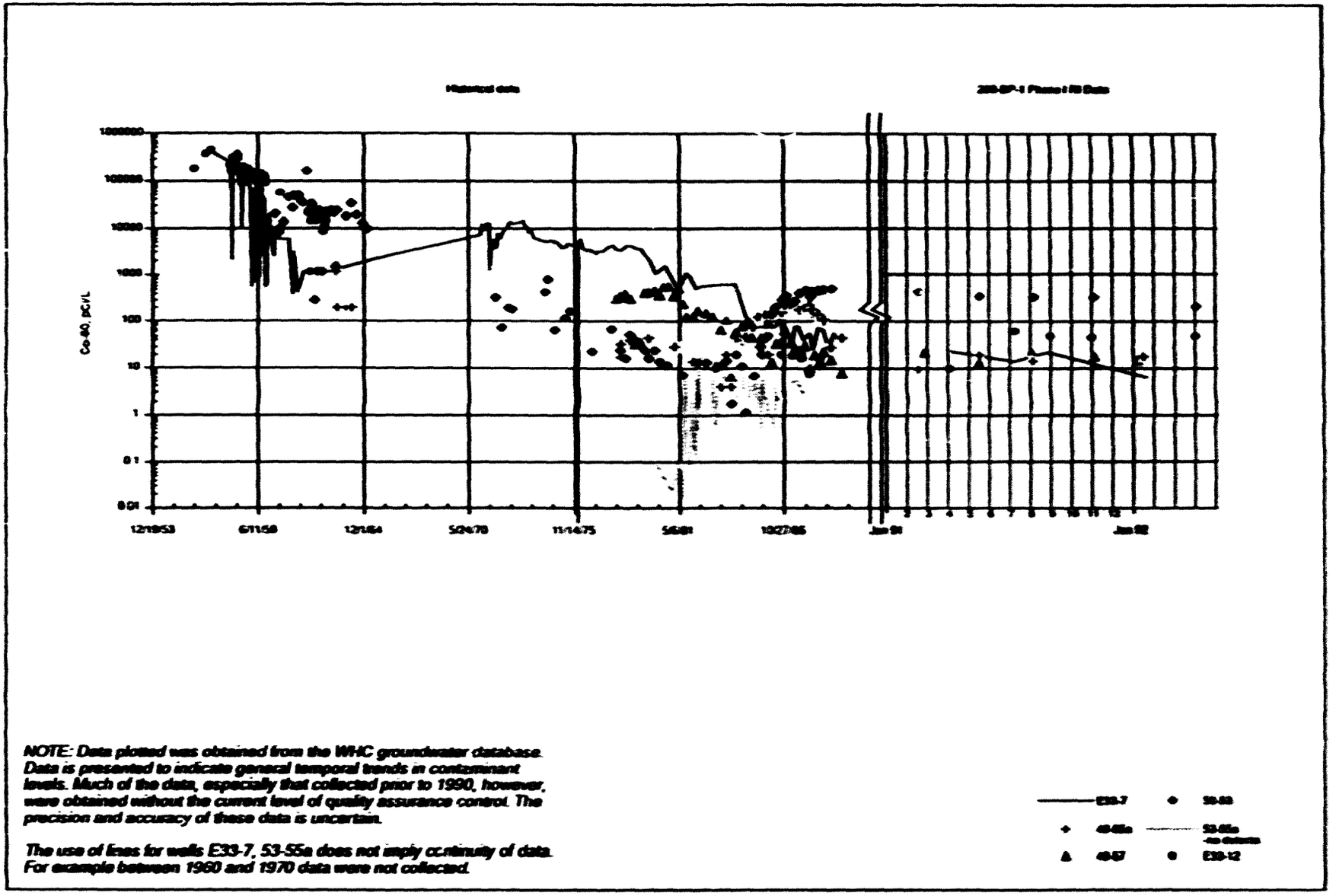


Figure 4-7. Historical Chromium Concentrations in Selected Wells.



913 1764/442403-10-93

Figure 4-8. Historical Co-60 Concentrations in Selected Wells.

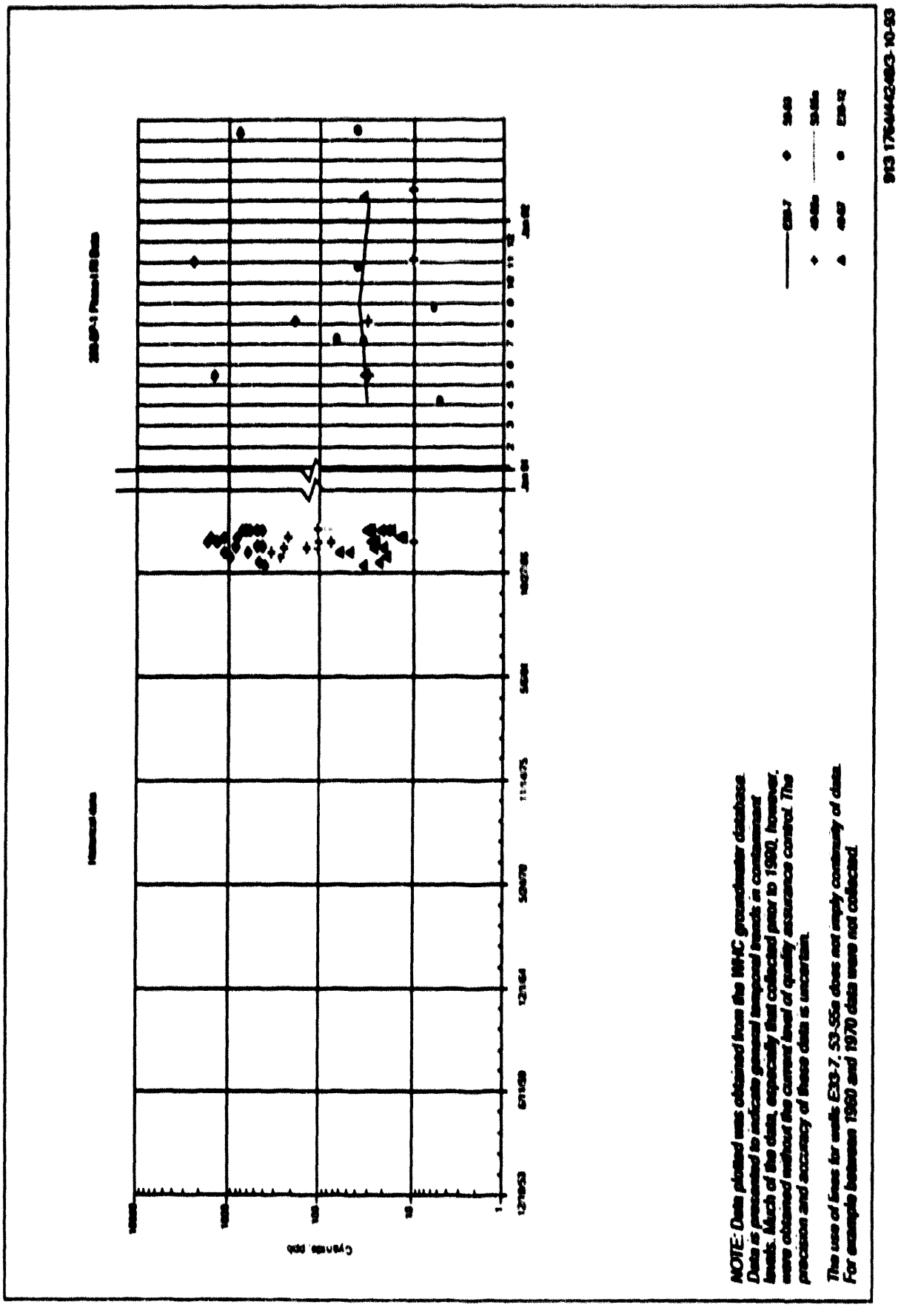


Figure 4-9. Historical Cyanide Concentrations in Selected Wells.

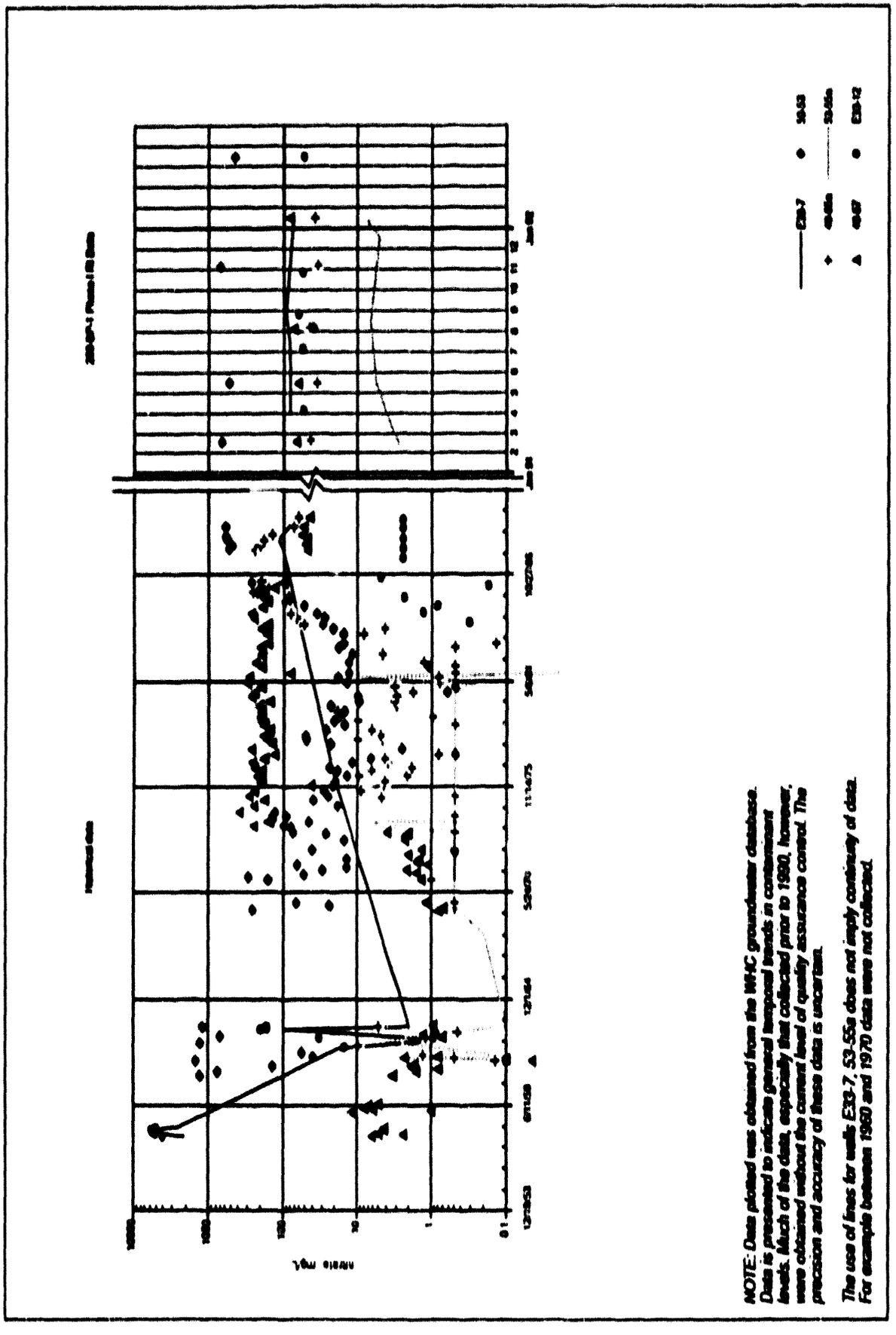


Figure 4-10. Historical Nitrate Concentrations in Selected Wells.

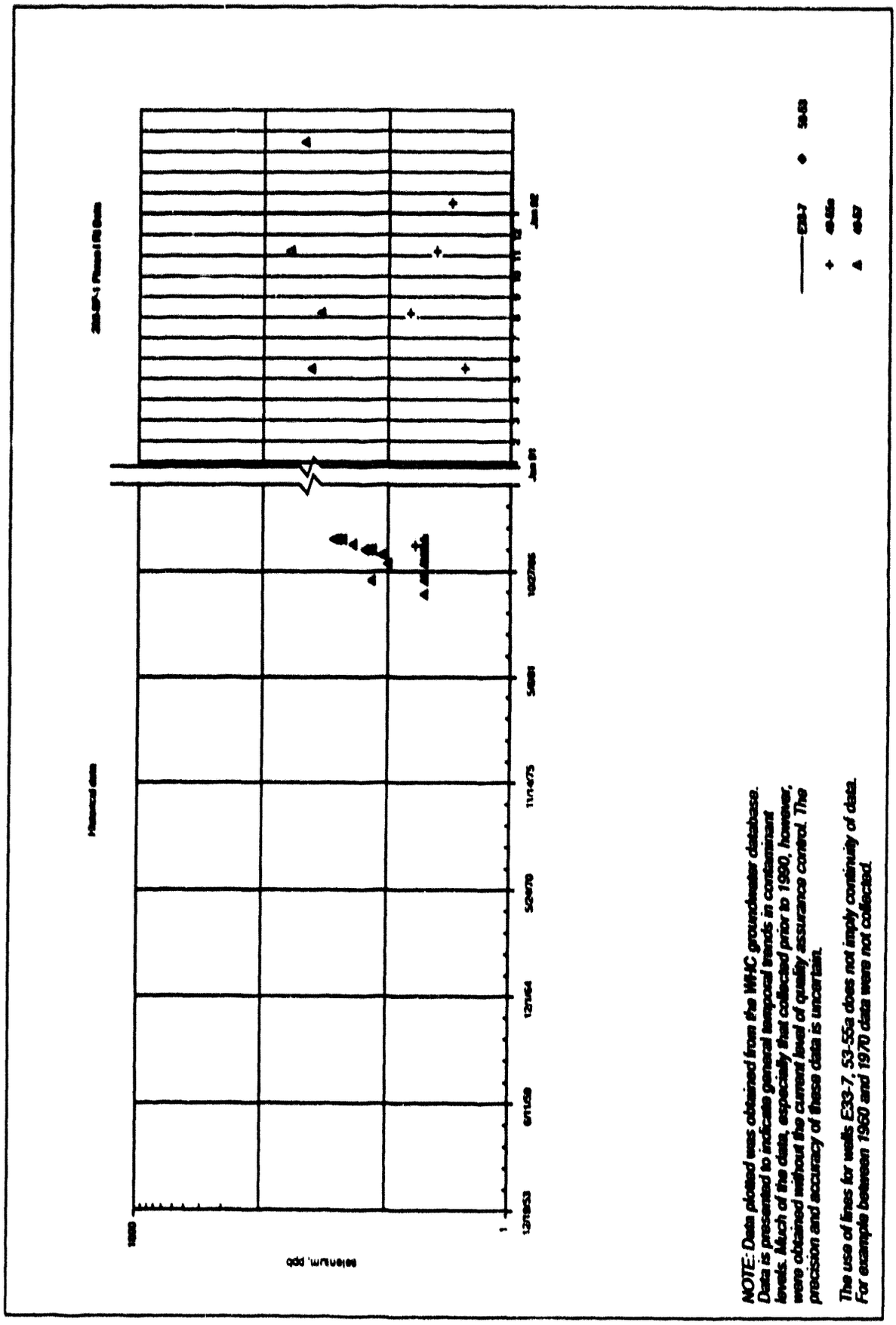
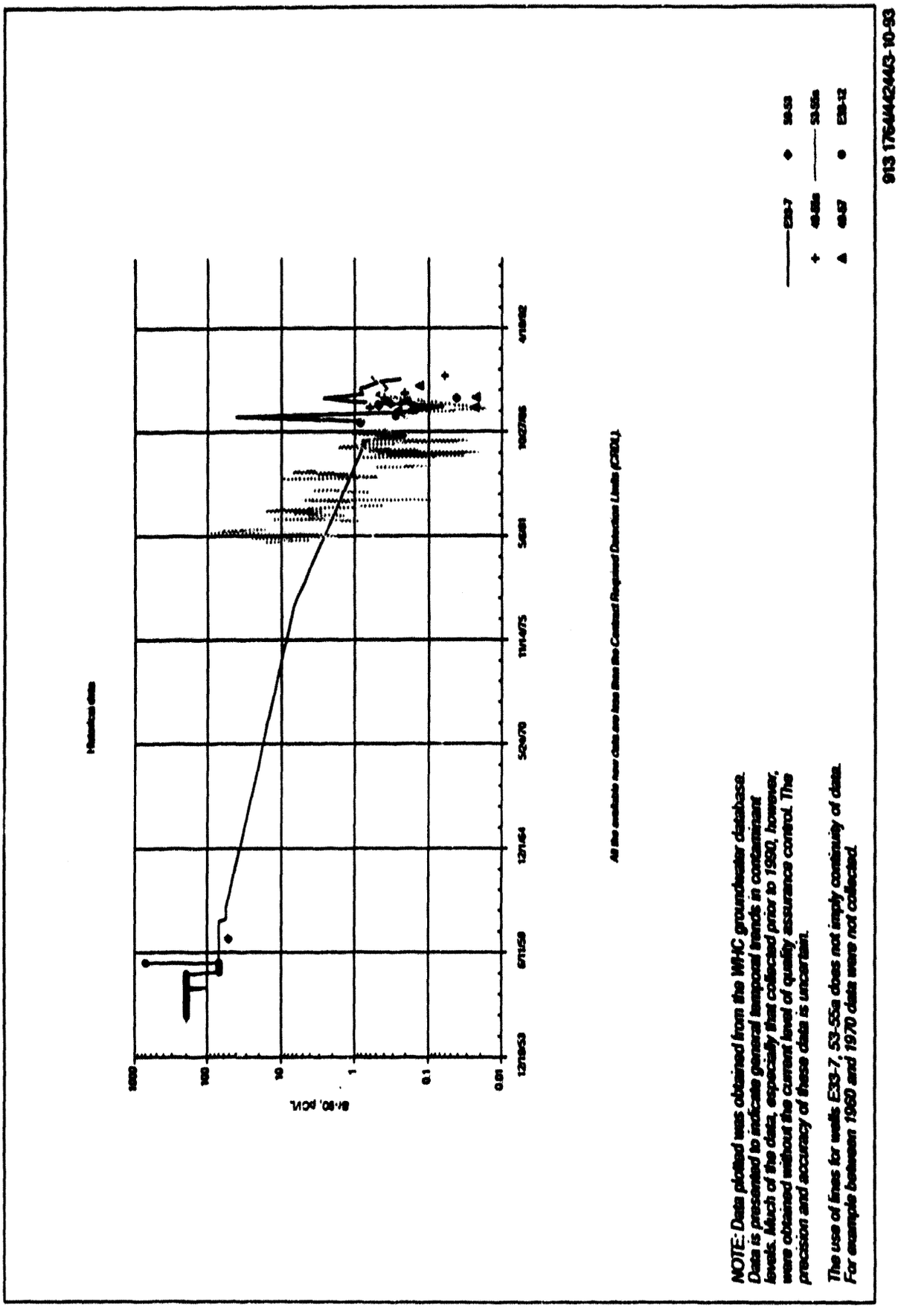


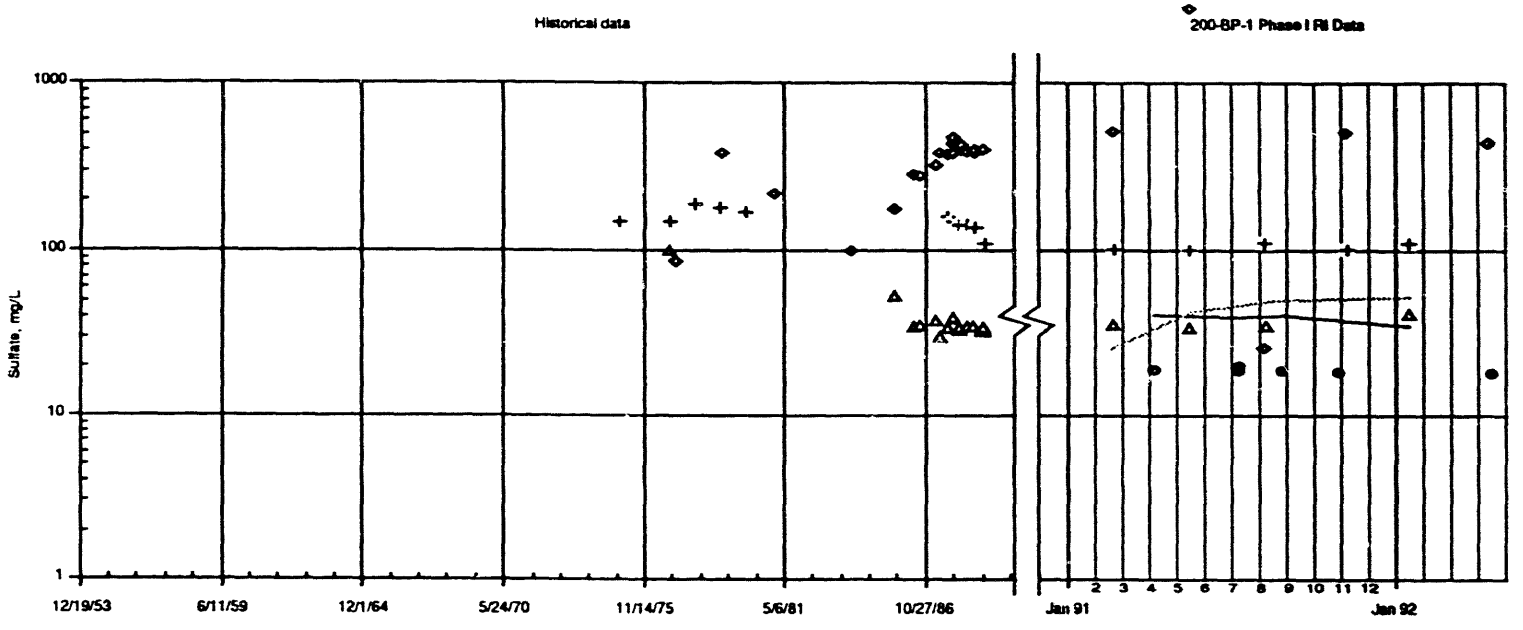
Figure 4-11. Historical Selenium Concentrations in Selected Wells.



NOTE: Data plotted was obtained from the WHC groundwater database. Data is presented to indicate general temporal trends in contaminant levels. Much of the data, especially that collected prior to 1980, however, were obtained without the current level of quality assurance control. The precision and accuracy of these data is uncertain.

The use of lines for wells E33-7, 53-55a does not imply continuity of data. For example between 1960 and 1970 data were not collected.

Figure 4-12. Historical Sr 90 Concentrations in Selected Wells.

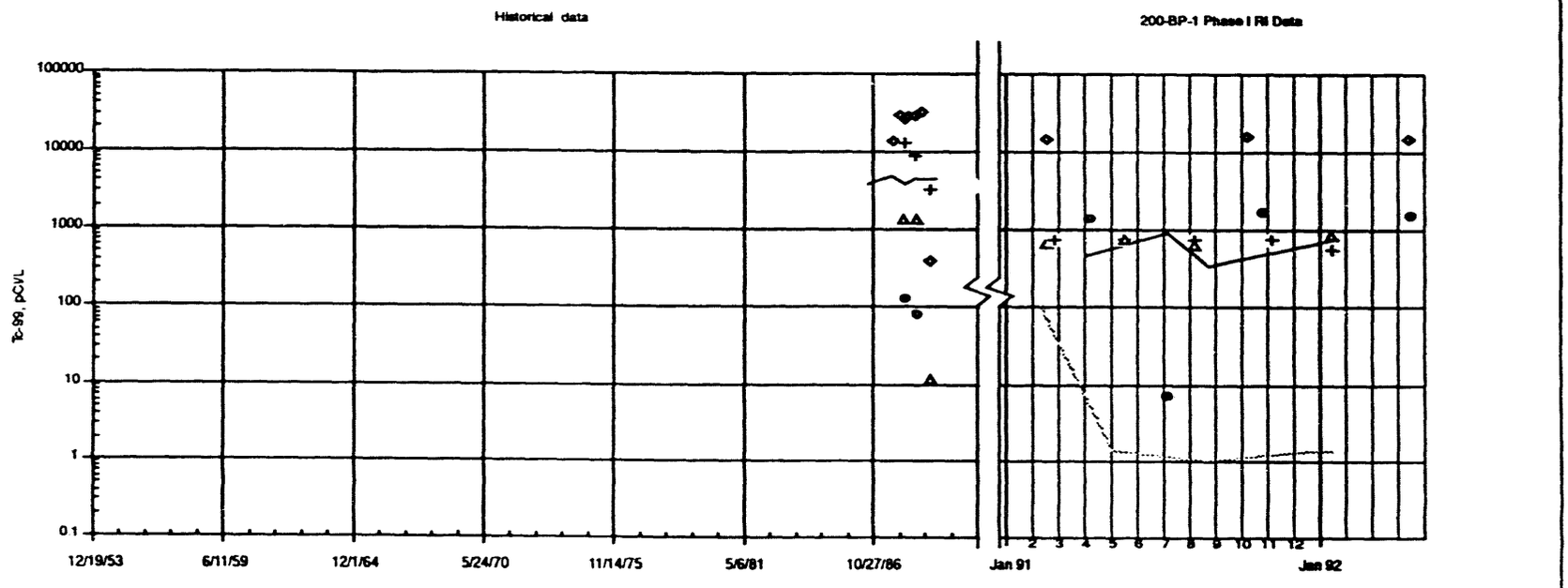


NOTE: Data plotted was obtained from the WHC groundwater database. Data is presented to indicate general temporal trends in contaminant levels. Much of the data, especially that collected prior to 1990, however, were obtained without the current level of quality assurance control. The precision and accuracy of these data is uncertain.

The use of lines for wells E33-7, 53-55a does not imply continuity of data. For example between 1960 and 1970 data were not collected.

913 1764/44243/3-10-93

Figure 4-13. Historical Sulfate Concentrations in Selected Wells.

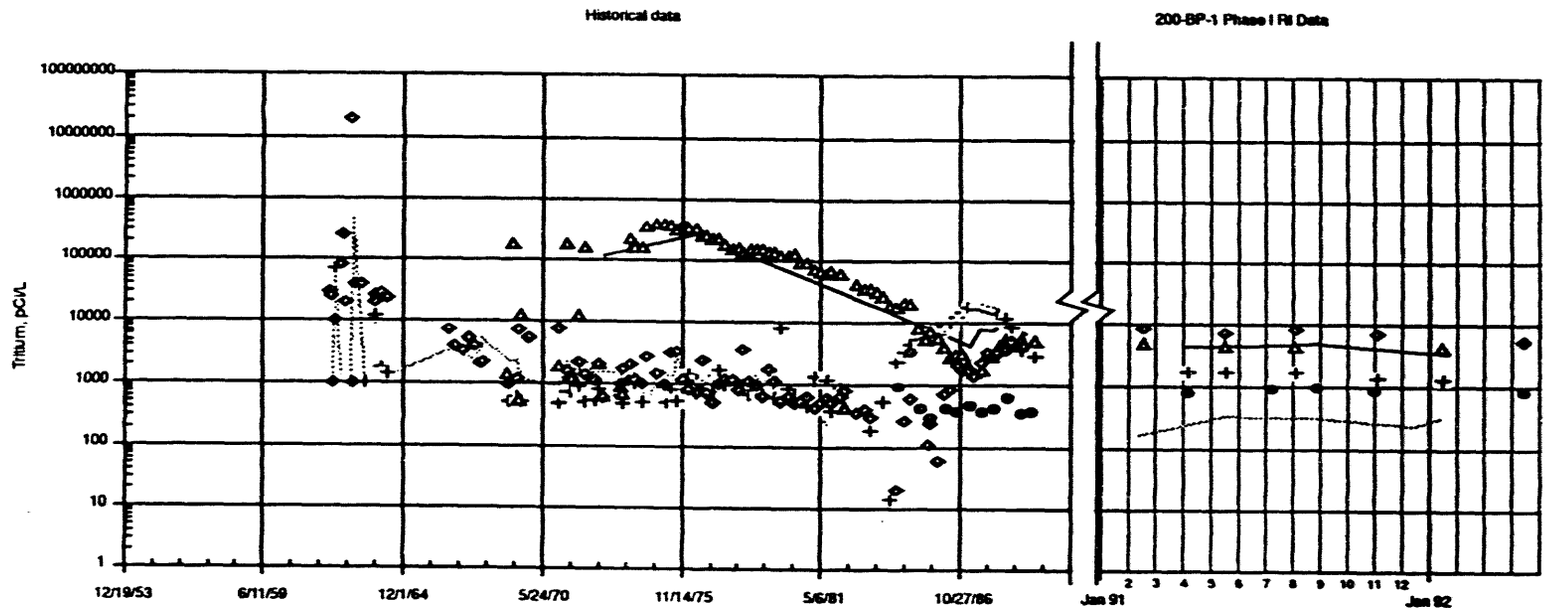


NOTE: Data plotted was obtained from the WHC groundwater database. Data is presented to indicate general temporal trends in contaminant levels. Much of the data, especially that collected prior to 1990, however, were obtained without the current level of quality assurance control. The precision and accuracy of these data is uncertain.

The use of lines for wells E33-7, 53-55a does not imply continuity of data. For example between 1960 and 1970 data were not collected.

913 1764/44245/3-10-93

Figure 4-14. Historical Tc-99 Concentrations in Selected Wells.

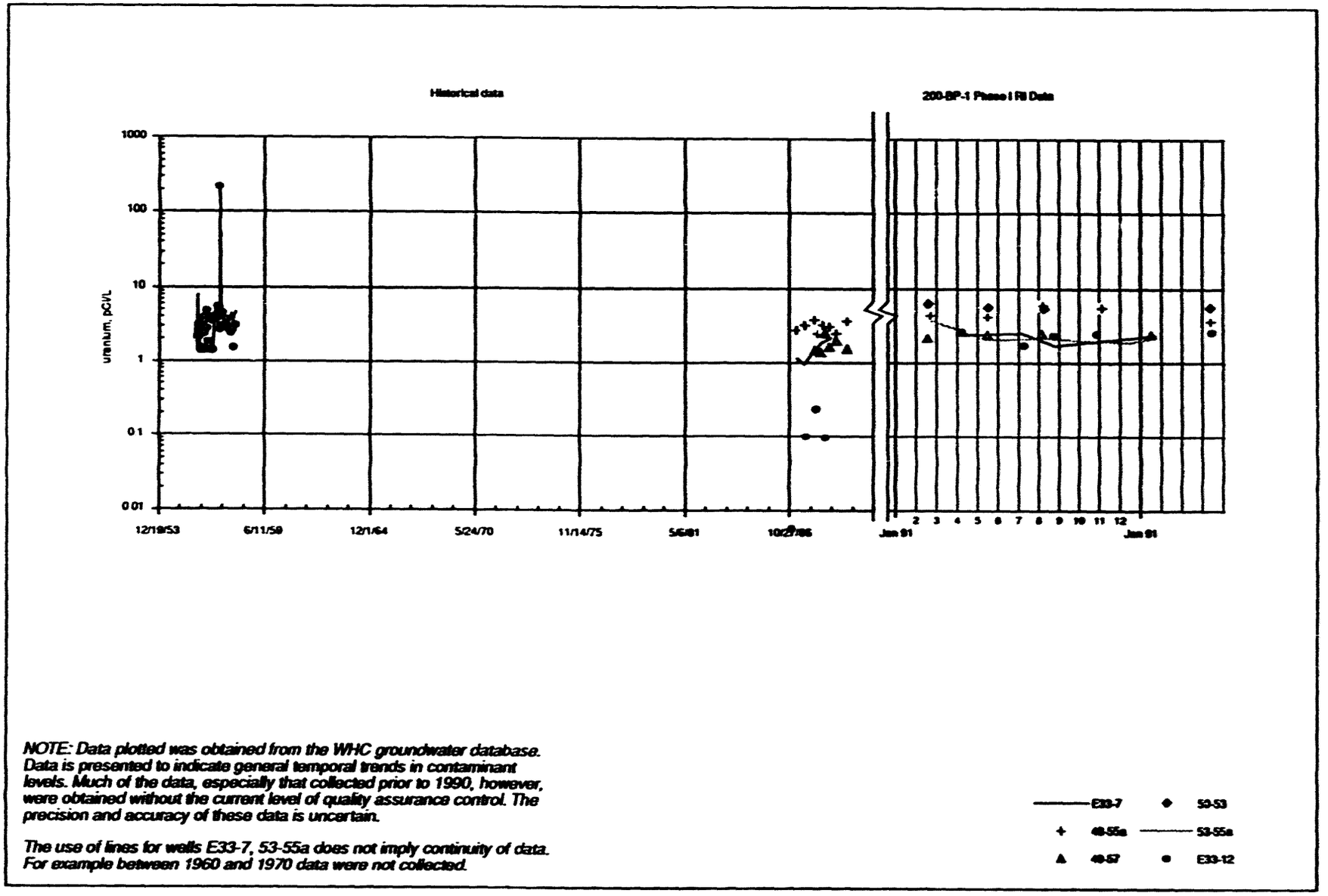


NOTE: Data plotted was obtained from the WHC groundwater database. Data is presented to indicate general temporal trends in contaminant levels. Much of the data, especially that collected prior to 1990, however, were obtained without the current level of quality assurance control. The precision and accuracy of these data is uncertain.

The use of lines for wells E33-7, 53-55a does not imply continuity of data. For example between 1960 and 1970 data were not collected.

913 1764/44246/3- 0-93

Figure 4-15. Historical Tritium Concentrations in Selected Wells.



913 1764/44247/3-10-93

Figure 4-16. Historical Uranium Concentrations in Selected Wells.

Table 4-1. Estimated Waste Quantities Discharged to Cribs in the 200-BP-1 Operable Unit.

	216-B-43	216-B-44	216-B-45	216-B-46	216-B-47	216-B-48	216-B-49	216-B-50	216-B-57	Total
Chemical Disposed (kg)										
Ferrocyanide	1,100	3,000	2,600	4,000	2,000	2,200	4,000			18,900
Ammonium Carbonate								9,100	12,000	21,100
Nitrate	400,000	800,000	90,000	1,200,000	700,000	1,000,000	1,500,000	1,500		5,691,500
Ammonium Nitrate								10,000		10,000
Phosphate	21,000	40,000	41,000	70,000	40,000	60,000	60,000			332,000
Sodium	170,000	330,000	340,000	500,000	310,000	400,000	600,000	500		2,650,500
Sulfate	29,000	60,000	60,000	100,000	60,000	80,000	80,000			469,000
Radionuclide Inventory (Ci)[a]										
H-3[b]										2,499
Co-60[b]										4490
Cs-137	1.3×10^{2}	3.09×10^{2}	6.66×10^{2}	8.89×10^{1}	6.66×10^{1}	2×10^{2}	1.82×10^{2}	5.12×10^{1}	2.26×10^{2}	1.92×10^{3}
Ru-106	6.9×10^{-1}	1.4×10^{-7}	4.68×10^{-7}	7.7×10^{-7}	5.23×10^{-7}	1.87×10^{-7}	6.6×10^{-7}	1.08×10^{-6}	4.76×10^{-6}	0.69
Sr-90	5.74×10^{2}	1.2×10^{3}	1.18×10^{3}	6.31×10^{2}	2.61×10^{2}	5.47×10^{2}	1.14×10^{3}	3.39×10^{0}	1.83×10^{0}	5.5×10^{3}
Plutonium (g)	5×10^{-1}	1.5×10^{1}	1×10^{1}	2×10^{1}	5×10^{0}	5×10^{0}	1.5×10^{1}	2.39×10^{-1}	1.87×10^{-1}	7.1×10^{1}
U-Gross	4.54×10^{-3}	7.56×10^{-4}	2.27×10^{-3}	6.35×10^{-2}	2.27×10^{-3}	7.57×10^{-4}	1.06×10^{-1}	9.51×10^{-6}	2.97×10^{-4}	1.8×10^{-1}
Alpha	3.07×10^{-2}	9.21×10^{-1}	6.14×10^{-1}	1.23×10^{0}	3.07×10^{-1}	3.07×10^{-1}	2.62×10^{2}	1.47×10^{-2}	1.15×10^{-2}	2.65×10^{2}
Beta	1.4×10^{3}	2.99×10^{3}	3.64×10^{3}	1.44×10^{3}	6.5×10^{2}	1.49×10^{3}	2.63×10^{3}	1.05×10^{2}	4.37×10^{2}	1.48×10^{4}
Other Suspected Waste[b]										
Constituents										
Tc-99										Unknown
Tributyl Phosphate										Trace
Paraffin Hydrocarbons										Trace

Source: DOE-RL (1992a).

[a]Curies decayed through 12/31/89, except for H-3 and Co-60 which are decayed through 4/1/86.
[b]Stenner et al. (1988). Data inventories were not provided on a by crib basis, and are reported as total waste quantities.

Table 4-2. Blank Adjustment Data for Soil Samples Analyzed by Weston Laboratory.

Parameter	Number of Detects	Maximum Value Detected (ppb)	Multiplication Factor[a]	Adjustment Number Used (ppb)	Number of Samples Blank Adjusted
Volatiles					
Toluene	1	4	10	40	19
Acetone	2	150	10	1500	12
Chloroform	1	13	5	65	0
1,1,1 Trichloroethane	1	1	5	5	0
Methylene chloride	2	49	10	490	15
2-Butanone	1	11	10	110	0
Semi-volatiles					
2-4 Dimethylphenol	1	5	5	25	0
4-Methylphenol	1	8	5	40	0
1,4-Dichlorobenzene	1	2	5	10	0
Phenol	1	2	5	10	0
2-Methyphenol	1	1	5	5	0
1,2-Dichlorobenzene	1	10	5	50	0
Bis(2-ethylhexyl)phthalate	1	1	10	10	0
Di-n-octyphthalate	1	2	10	20	0
Butybenzylphthalate	1	2	10	20	0
Pesticides					
Heptachlor epoxide	1	0.01	5	0.05	0
4,4'-DDT	1	0.17	5	0.85	0
Radionuclides		(pCi/L)		(pCi/ml)	
Potassium-40	3	263	5	1.31	0
Plutonium-238	1	1.8	5	0.009	0
Strontium-90	3	2	5	0.01	0
Technetium-99	1	1.8	5	0.009	0
Tritium	4	230	5	1.15	0
Plutonium-239	1	0.01	5	0.00005	0
Thorium-228	1	10	5	0.05	0
Gross Alpha	1	2	5	0.01	0
Gross Beta	4	41	5	0.205	0

[a]Reference: Bleyler 1988a,b

Table 4-3. Background UTLs for Soils and Groundwater.[a] (Sheet 1 of 2)

Parameter	Operable Unit Specific Background				Hanford Site Background	
	Soils				Soil	Groundwater
	X	s	n	UTL[b] mg/kg	UTL[c] mg/kg	UTL[d] µg/L
Inorganics						
aluminum	4,583	1,216	3	13,892	15600	<200
antimony	.*	.*	3	9.1*	--	--
arsenic	2.0	0.6	3	6.7	8.92	10
barium	69.0	7.0	3	123	171	68.5
beryllium	0.4	0.1	3	1.1	1.77	<5
bismuth	.*	.*	14	10*	--	<5
cadmium	.*	.*	3	0.7*	--	<10
calcium	7,993	2,203	3	24,857	23,920	63,000
chromium	1.6	1.1	3	10.0	27.9	<30
cobalt	6.7	1.3	3	17	19.6	.
copper	10.9	2.3	3	28.2	28.2	<30
cyanide (total)	0.18	0.2	16	0.7	--	--
cyanide (free)	.*	.*	4	0.2*	--	--
iron	11,900	2,425	3	30,463	39,160	86^{L} 291^{M} 818^{H}
lead	3.6	0.46	3	7.1	14.75	<5
magnesium	3,437	891	3	10,260	8,760	16,480
manganese	224	53	3	630	612	24.5^{L} 163.5^{H}
mercury	.*	.*	3	0.1*	1.25	<0.1
nickel	7.3	2.4	3	25.7	25.3	<30
potassium	1,033	374	3	3,898	3,120	7,975
selenium	0.2	0.06	17	0.35	--	<5
silicon	--	--	--	--	--	26,500
silver	0.98	0.43	3	4.3	2.7	<10
sodium	220	138	3	1,273	1,290	33,500
strontium	--	--	--	--	--	264.1
thallium	--	--	3	0.3*	--	--
vanadium	15.2	5.8	3	59.6	111	15

Table 4-3. Background UTLs for Soils and Groundwater.[a] (Sheet 2 of 2)

Parameter	Operable Unit Specific Background				Hanford Site Background	
	Soils				Soil	Groundwater
	X	s	n	UTL[b] mg/kg	UTL[c] mg/kg	UTL[d] µg/L
zinc	27.2	6.4	3	76.2	79	<50[L] 673[H]
ammonia	--	--	--	--	28.2	120
chloride	23.9	4.7	3	60	763	8,690[L] 28,500[H]
fluoride	3.1	2.1	3	19.2	12	775[L] 1,340[H]
nitrate	1.2	1.2	17	4.2	199	12,400
nitrite	.*	.*	17	1.4*	--	--
phosphate	.*	.*	17	3*	16	<1,000
sulfate	13	9.1	17	35.5	1320	90,500
Naturally-Occurring Radionuclides				pCi/g		pCi/L
Gross alpha	3.22	2.75	18	10.2	--	5.79[L] 63[H]
Gross beta	7.24	11.75	18	37.1	--	12.62[L] 35.5[H]
Potassium-40	11.4	1.4	4	18.5	--	--
Radium-226	0.58	0.39	4	2.6	--	0.23e
Thorium-228	0.56	0.04	4	0.76	--	--
Total Uranium	0.32	0.04	16	0.42	--	3.43 (2.64 µg/L)

[a]Background values are assumed only for TAL parameters and K-40, RA-226 and total uranium. Background values for all other analytes (TCL parameters and fission products) are assumed to be 0.
[b]Value calculated from operable unit specific background.
[c]Value from Hanford Site Soil Background (DOE-RL 1993a).
[d]Value from Hanford Site Groundwater Background (DOE-RL 1992b).
[e]Represents value for radium.
*Parameter was never detected in the respective background samples; therefore, the highest reported background sample quantitation limit (SQL) is substituted as a surrogate UTL.

L = Low (See Section 4.2.2.2)
M = Medium (See Section 4.2.2.2)
H = High (See Section 4.2.2.2)
-- = Not available or not applicable.
X = Mean.
s = standard deviation.
n = number of samples.
UTL = upper 95 percentile tolerance limit.

Table 4-4. Results of Background Screening in Soils for Inorganic TAL Parameters and Naturally Occuring Radionuclides.

Below Background			Above Background		
Parameter	Maximum Concentration (mg/kg)	Background (mg/kg)	Parameter	Maximum Concentration (mg/kg)	Background (mg/kg)
antimony	5.8	9.1*	aluminum	15800	15600
arsenic	8.2	8.92	bismuth[a]	41.9	10*
barium	129	171	cadmium	3.9	0.7*
beryllium	0.7	1.77	cyanide (total)	249	0.7
calcium	12,500	23,920	cyanide (free)	4.1	0.2*
cobalt	13	19.6	chromium	119.0	27.9
iron	36,200	39,160	copper	76.9	28.2
mercury	0.4	1.25	potassium	11,600	3,120
manganese	464	612	magnesium	24,200	6,760
silver	2.4	2.7	sodium	9500	1,290
vanadium	99.8	111	nickel	251	25.3
zinc	59.9	79	lead	27.7	14.75
chloride	79.2	763	thallium	0.6	0.3*
fluoride	6.7	12	selenium	0.46	0.35
sulfate	1080	1320	nitrate	5470	199
			nitrite	109	1.4*
			phosphate	154	16
			potassium-40	18.7 (pCi/g)	18.5(pCi/g)
			radium-226	15.2 (pCi/g)	2.6*(pCi/g)
			total U	341 (pCi/g)	0.42(pCi/g)
			thorium-228	1.17 (pCi/g)	0.76(pCi/g)
			Gross alpha	9279	10.2(pCi/g)
			Gross beta	39500000	37.1(pCi/g)

*Parameter was never detected in the respective background samples; therefore, the highest reported background sample quantitation (SQL) is substituted as a surrogate UTL.
[a]Bismuth is not a TAL parameter, however it is included as it was analyzed due to site contamination history.

Table 4-5. Blank Adjustment Data for Groundwater Samples Analyzed by Weston Laboratory. (Sheet 1 of 2)

Quarter	Parameter	Number of Detects	Maximum Value Detected (ug/L)	Multiplication Factor[a]	Adjustment Number Used (ug/L)	Number of Samples Blank Adjusted
Inorganics						
1	cobalt	1	6.3	5	31.5	0
1	magnesium	1	98.9	5	494.5	0
1	manganese	1	2.0	5	10.0	28
1	sodium	1	89.7	5	448.5	0
1	nickel	1	9.1	5	45.5	27
1	selenium	1	1.0	5	5.0	38
1	vanadium	1	8.3	5	41.5	3
1	zinc	3	9.5	5	47.5	20
2	copper	2	6.6	5	33.0	14
2	mercury	1	1.1	5	5.5	8
2	nickel	1	14.0	5	70.0	19
2	selenium	1	2.1	5	10.5	30
2	zinc	1	6.0	5	30.0	0
4	barium	1	36.5	5	182.5	62
4	calcium	1	186000	5	930000	70
4	copper	1	11.7	5	58.5	12
4	magnesium	1	3960	5	19800	67
4	sodium	1	1770	5	8850	1
5	silver	2	4.5	5	22.5	1
5	copper	1	26.2	5	131	5
5	lead	2	8.7	5	43.5	4
5	zinc	2	8.7	5	43.5	6
Anions						
2	chloride	2	6.0	5	30.0	41
2	fluoride	1	0.5	5	2.5	33
2	sulfate	1	0.3	5	1.5	0
4	chloride	1	0.6	5	3.0	0
4	nitrate	1	0.5	5	2.5	1
4	sulfate	1	9.4	5	47.0	24

Table 4-5. Blank Adjustment Data for Groundwater Samples Analyzed be Weston Laboratory. (Sheet 2 of 2)

Quarter	Parameter	Number of Detects	Maximum Value Detected (ug/L)	Multiplication Factor[a]	Adjustment Number Used (ug/L)	Number of Samples Blank Adjusted
Organics						
Volatiles						
1	4-methyl-2-pentanone	2	89.0	5	445.0	3
1	toluene	1	1.0	10	10.0	0
1	chloromethane	1	2.0	5	10.0	1
1	chloroform	18	5.0	5	25.0	0
1	carbon disulfide	1	1.0	5	5.0	1
2	chloroform	1	1.0	5	5.0	0
3	acetone	2	12.0	10	120.0	0
4	methlylene chloride	1	55.0	10	550.0	0
5	methylene chloride	1	41.0	10	410.0	0
5	toluene	1	1.0	10	10.0	0
Semivolatiles						
1	benzyl alcohol	1	3.0	5	15.0	0
1	bis (2-ethylhexyl) phthalate	2	2.0	10	20.0	13
1	di-n-butylphthalate	1	2.0	10	20.0	10
1	n-nitrosodiphenylamine	1	1.0	5	5.0	8
Radionuclides						
			(pCi/L)		(pCi/L)	
1	technetium-99	1	4	5	20	4
3	gross beta	1	36	5	180	24
3	strontium-90	2	1	5	5	1
4	tritium	1	230	5	1150	6
5	gross beta	1	14	5	70	24
5	tritium	2	1500	5	7500	34

[a]Reference: Bleyler 1988a and b.

Table 4-6. Results of Background Screening in Groundwater for TAL Parameters and Naturally Occurring Radionuclides. (Sheet 1 of 2)

Below Background			Above Background		
Parameter	Maximum Concentration (µg/L)	Background[a] (µg/L)	Parameter	Maximum Concentration (µg/L)	Background[a] (µg/L)
zinc	183	<50[L] 673[H]	aluminum	244	<200
phosphate	250	<1000	antimony	120	--
			arsenic	10.9	10
			barium	216	68.5
			beryllium	10.7	<5
			bismuth[b]	258	<5
			calcium	289,000	63,000
			cadmium	16.7	<10
			cyanide (total)	2,710	--
			cobalt	3.5	--
			chromium	56.4	<30
			copper	107	<30
			iron	1,180	86[L] 291[M] 818[H]
			lead	84.1	<5
			magnesium	77,800	16,480
			manganese	295	24.5[L] 163.5[H]
			mercury	0.4	<0.1
			nickel	701	<30
			potassium	51,100	7,975
			selenium	53.1	<5
			silicon	29,690	26,500
			silver	61.3	<10
			sodium	63,300	33,500
			strontium	1,420	264.1
			tin	118	--
			thallium	2.2	--
			vanadium	76.3	15
			ammonia	260	120
			chloride	79,700	8,690[L] 28,500[H]
			fluoride	1,900	775[L] 1,340[H]
			nitrate	1,970,000	12,400
			sulfate	2,930,000	90,500
			nitrite	1,200	--
			(pCi/L)		
			gross alpha	74	5.79[L] 63[H]
			gross beta	7100	12.62[L] 35.5[H]
			potassium-40	394	--
			radium-226	262	0.23

Table 4-6. Results of Background Screening in Groundwater for TAL Parameters and Naturally Occurring Radionuclides. (Sheet 2 of 2)

Below Background			Above Background		
Parameter	Maximum Concentration (µg/L)	Background[a] (µg/L)	Parameter	Maximum Concentration (µg/L)	Background[a] (µg/L)
			total uranium	14(µg/L)	2.64(µg/L) 3.43(pCi/L)
			radium-228	1.1	0.23

[a]Source: DOE-RL 1992b.
[b]Bismuth is not a TAL parameter, however it is included it was analyzed for due to site contamination history.
-- = Not available
L = Low (see Section 4.2.2.2)
M = Medium (see Section 4.2.2.2)
H = High (see Section 4.2.2.2)
TAL = Target Analyte List

Table 4-7. Preliminary Risk-Based Screening Exposure Factors.

Exposure Factor	HSBRAM Reasonable Maximum Exposure[a]
Intake Rate - Soil Ingestion	
Adult	100 mg/d
Child	200 mg/d
Intake Rate - Groundwater Ingestion	
Adult	2 L/d
Child	1 L/d
Intake Rate - Inhalation	20 m^3/d
Adult	10 m^3/d
Child	
Intake Rate - External Exposure	24 hr x 4E-04 yr/hr
Exposure Frequency	365 d/yr
Exposure Duration	
Adult	24 yr
Child	6 yr
Total	30 yr
Body Weight	
Adult	70 kg
Child	16 kg
Averaging Time	
Carcinogens	70 yr x 365 d/yr
Non-Carcinogens	6 yr x 365 d/yr or 30 yr x 365 d/yr
Particulate Emission Factor	2E+07 m^3/kg[b]

[a]DOE-RL 1993b; Factors based on EPA-10 (1991) and WAC 173-340.
[b]Calculated factor; see section 4.3.1.2.2.1

Table 4-8. Contaminants Carried Through The Risk-Based Screening Process for 200-BP-1 Operable Unit Soils. (Sheet 1 of 3)

Near Surface Soil Contaminants				Subsurface Infiltration Gravels/Soils Contaminants			
Contaminants	Maximum Concentration	Number of Detects	Number of Analyses[a]	Contaminants	Maximum Concentration	Number of Detects	Number of Analysis[a]
Inorganic Contaminants	mg/kg				mg/kg		
				cadmium	3.9	17	82
cadmium	1.6	9	53	chromium	119	80	82
chromium	11.8	48	53	copper	44	75	82
copper	76.9	53	53	cyanide (complex)	247	21	21
cyanide (free)	1.4	1	3	cyanide (free)	4.1	22	22
cyanide (total)	0.2	1	107	cyanide (total)	249	54	135
lead	27.7	52	52	lead	24.3	81	82
nickel	16.6	52	53	nickel	251	82	82
nitrate	223	68	68	thallium	0.5	27	82
selenium	0.3	4	57	nitrate	5,470	82	82
thallium	0.6	2	53	nitrite	109	26	56
				secenium	0.5	3	82
Organic Contaminants	mg/kg			Organic Contaminants	mg/kg		
carbon disulfide	0.001	1	53	acetone	0.083	23	76
methylene chloride	0.058	14	55	2-hexanone	0.001	1	76
benzoic acid	0.058	2	50	4-methyl 2-pentanone	0.011	2	76
bis(2-ethyhexyl)phthalate	0.28	17	48	styrene	0.001	1	76
butylbenzylphthalate	1.3	2	51	toluene	0.002	3	74
2-chloronapthalene	0.074	2	52	1,1,1-trichloroethane	0.01	6	75
chrysene	0.04	1	52	benzoic acid	0.7	4	77

Table 4-8. Contaminants Carried Through The Risk-Based Screening Process for 200-BP-1 Operable Unit Soils. (Sheet 2 of 3)

Near Surface Soil Contaminants				Subsurface Infiltration Gravels/Soils Contaminants			
Contaminants	Maximum Concentration	Number of Detects	Number of Analyses[a]	Contaminants	Maximum Concentration	Number of Detects	Number of Analysis[a]
di-n-butlyphthalate	3.1	11	52	bis(2-ethyhexyl)phthalate	2.7	31	76
pentachlorophenol	0.15	3	52	butylbenzylphthalate	1.3	11	76
phenol	0.12	1	52	2-chlorophenol	0.008	1	78
pyrene	0.049	1	52	diethylphthalate	0.620	4	78
4,4'-DDT	0.011	3	51	di-n-butylphthalate	2.5	14	79
PCB	0.022	1	52	di-n-octylphthalate	0.270	18	78
				isophorone	0.190	1	78
				pentachlorophenol	0.2	4	78
				tributyl phosphate	94	26	27
				4,4'DDT	0.002	1	77
				endosulfan II	0.017	1	77
				PCB	0.77	3	77
				aldrin	0.007	1	77
Radionuclide Contaminants	pCi/g			Radionuclide Contaminants	pCi/g		
cesium-137	265	75	77	antimony-125	50.4	10	10
manganese-54	0.07	1	79	cesium-137	12900000	55	75
plutonium-238	0.044	3	62	chromium-51	56.6	2	2
plutonium-239	0.016	5	62	cobalt-60	62	32	82
potassium-40	18.7	79	79	plutonium-238	203	32	75
radium-226	15.2	65	79	plutonium-239	0.93	7	41

Table 4-8. Contaminants Carried Through The Risk-Based Screening Process for 200-BP-1 Operable Unit Soils. (Sheet 3 of 3)

Near Surface Soil Contaminants				Subsurface Infiltration Gravels/Soils Contaminants			
Contaminants	Maximum Concentration	Number of Detects	Number of Analyses[a]	Contaminants	Maximum Concentration	Number of Detects	Number of Analysis[a]
strontium-90	13	52	66	plutonium-239+240	5850	33	33
technetium-99	1.7	5	53	potassium-40	16.9	44	44
thorium-228	0.94	77	79	radium-226	3.7	24	44
total-U	3.1	12	76	strontium-90	14200000	57	72
				technetium-99	210	33	62
				thorium-228	1.17	40	44
				total-U	341	45	76
				tritium	194.2	34	36

[a] Does not include data rejected during validation.

Table 4-9. Chemical Specific Constants Used for Calculation of Volatilization Factors and the Calculated Volatilization Factors.

Parameter	Henry's Law Constant[a] (atm-m^3mol)	Molecular Diffusivity[b] (cm^2/Sec)	Organic Carbon Coefficient (Koc) (cm^3/g)	Vapor Pressure[a] (mm Hg)	Molecular Weight	Volatilization Factor (VF) (m^3/kg)
2-Hexanone	2.7E-05[d]	9.5E-02[d]	4.5E+00[ad]	4.00E+00	100	1.5E+04
Methylene Chloride	2.0E-03	1.08E-01	8.8E+00[a]	3.62E+02	85	1.6E+03
4-Methyl 2-Pentanone	1.5E-04	7.8E-02	1.9E+01[c]	2.0E+01	100	1.0E+04
Styrene	2.5E-03	7.5E-02	2.7E+02[b]	4.5E+00	104	9.8E+03
1,1,1-Tricholoethane	1.44E-02	8.4E-02	1.5E+02[a]	3.0E+01	133	2.8E+03
Carbon Disulfide	1.2E-02	1.0E-01	5.4E+01[a]	3.6E+02	76	1.6E+03

[a]EPA (1990)
[b]EPA (1988c)
[c]Howard (1990)
[d]2-Butanone was used as a surrogate for 2-Hexanone because data was unavailable.

Table 4-10. Preliminary Risk-Based Screening for Non-Radioactive Soil Contaminants, In Near Surface Soils (0-15 feet), at the 200-BP-1 Operable Unit. (Sheet 1 of 2)

Parameter	Maximum Detected Soil Concentration (mg/kg)	Oral RfD (mg/kg-d)	Soil Concentration at Oral HQ=0.1 (mg/kg)	Inhalation RfD (mg/kg-d)	Soil Concentration at Inhalation HQ=0.1 (mg/kg)	Oral SF $(mg/kg\text{-}d)^{-1}$	Soil Concentration at Oral ICR = 1E-07 (mg/kg)	Inhalation SF $(mg/kg\text{-}d)^{-1}$	Soil Concentration at Inhalation ICR = 1E-07 (mg/kg)	Regulatory Soil Cleanup Guidelines (mg/kg)
Inorganics										
cadmium	1.6	1.0E-03[a]	8	–[k]	–[k]	–[l]	–[l]	6.3E+00[a]	2.5	–
copper	76.9	4.0E-02[d]	320	–[k]	–[k]	–[l]	–[l]	–[l]	–[l]	–
cyanide (total)	0.2	2.0E-02[a,f]	160	–[k]	–[k]	–[l]	–[l]	–[l]	–[l]	–
lead	27.7	ND	–	ND	–[k]	ND	–[k]	ND	–[k]	250[j], 500-1,000[c]
chromium	11.8	5.0E-03	40	–[k]	–[k]	–[l]	–[l]	4.1E+01	0.4	–
nickel	16.6	2.0E-02	160	–[k]	–[k]	–[l]	–[l]	–[m]	–[m]	–
selenium	0.3	5.0E-03	40	–[k]	–[k]	–[l]	–[l]	–[l]	–[l]	–
cyanide (free)	1.4	2.0E-02	160	–[k]	–[k]	–[l]	–[l]	–[l]	–[l]	–
thallium	0.6	9.0E-05[a,g]	0.72	–[k]	–[k]	–[l]	–[l]	–[l]	–[l]	–
Anions										
nitrate	223	1.6E+00[a]	13,000	–[k]	–[k]	–[l]	–[l]	–[l]	–[l]	–
Volatile Organics										
carbon disulfide	0.001	1.0E-01[a]	800	2.9E-03[b]	0.76	–[l]	–[l]	–[l]	–[l]	–
methylene chloride	0.058	6.0E-02[a]	480	–[k]	–[k]	7.5E-03[a]	8.5	1.6E-03[a]	0.8[i]	–
Semi-Volatile Organics										
benzoic acid	0.058	4.0E+00[a]	32,000	–[k]	–[k]	–[l]	–[l]	–[l]	–[l]	–
bis(2-ethyhexyl)phthalate	0.28	2.0E-02[a]	160	–[k]	–[k]	1.4E-02[a]	4.6	–[k]	–[k]	–
butylbenzyphthalate	1.3	2.0E-01[a]	1,600	–[k]	–[k]	–[k]	–[k]	–[k]	–[k]	–
2-chloronaphthalene	0.074	8.0E-02[a]	640	–[k]	–[k]	–[l]	–[l]	–[l]	–[l]	–
chrysene	0.040	–[k]	–[k]	–[k]	–[k]	7.3E+00[a,h]	0.0068	–[k]	–[k]	–

Table 4-10. Preliminary Risk-Based Screening for Non-Radioactive Soil Contaminants, In Near Surface Soils (0-15 feet), at the 200-BP-1 Operable Unit. (Sheet 2 of 2)

Parameter	Maximum Detected Soil Concentration (mg/kg)	Oral RfD (mg/kg-d)	Soil Concentration at Oral HQ=0.1 (mg/kg)	Inhalation RfD (mg/kg-d)	Soil Concentration at Inhalation HQ=0.1 (mg/kg)	Oral SF $(mg/kg\text{-}d)^{-1}$	Soil Concentration at Oral ICR = 1E-07 (mg/kg)	Inhalation SF $(mg/kg\text{-}d)^{-1}$	Soil Concentration at Inhalation ICR = 1E-07 (mg/kg)	Regulatory Soil Cleanup Guidelines (mg/kg)
di-n-butylphthalate	3.1	1.0E-01[a]	800	_[k]	_[k]	_[l]	_[l]	_[l]	_[l]	_
pentachlorophenol	0.150	3.0E-02[a]	240	_[k]	_[k]	1.2E-01[a]	0.533	_[k]	_[k]	_
phenol	0.120	6.0E-01[a]	4,800	_[k]	_[k]	_[l]	_[l]	_[l]	_[l]	_
pyrene	0.049	3.0E-02[a]	240	_[k]	_[k]	_[l]	_[l]	_[l]	_[l]	_
Pesticides										
4,4'-DDT	0.011	5.0E-04[a]	4	_[k]	_[k]	3.4E-01[a]	0.18	3.4E-01[a]	48	_
PCB	0.022	_[k]	_[k]	_[k]	_[k]	7.7E+00[a]	0.0083	_[k]	_[k]	1-25[e]

[a]Integrated Risk Information System (IRIS, EPA 1993)
[b]Health Effects Assessment Summary Tables (HEAST, EPA 1991c or EPA 1992b)
[c]EPA 1989a
[d]Superfund Technical Support Center (STSC, EPA 1991b)
[e]40 CFR 761
[f]Based on toxicity factor for free cyanide
[g]Oral RfD based on thallium-acetate
[h]Surrogate oral SF based on Benzo(a)pyrene
[i]Value based on inhalation of volatiles
[j]MTCA guidelines
[k]No RfD or SF available to evaluate this pathway
[l]Not classified as a carcinogen or not carcinogenic via this exposure route or pathway
[m]Nickel is only carcinogenic as refinery dust
-- Indicates not available
HQ = Hazard quotient
ICR = Lifetime incremental cancer risk
RfD = chronic reference dose
SF = Slope factor
ND = Not determined
Note: Shaded areas indicate screening criterion exceeded

Table 4-11. Preliminary Risk-Based Screening for Radioactive Soil Contaminants, In Near Surface Soils (0-15 feet), at the 200-BP-1 Operable Unit.

Parameter	Maximum Soil Concentration (pCi/g)	External SF[a] $(pCi\text{-}yr/g)^{-1}$	Soil Concentration at External ICR = 1E-07 (pCi/g)	Oral SF[a] $(pCi)^{-1}$	Soil Concentration at Oral ICR = (pCi/g)	Inhalation SF[a] $(pCi)^{-1}$	Soil Concentration at Inhalation ICR = 1E-07 (pCi/g)
cesium-137	265	2.0E-06	0.0017	2.8E-11	2.7	1.9E-11	480
manganese-54	0.07	2.9E-06	0.001	1.1E-12	69.1	5.3E-12	1,700
plutonium-238	0.044	2.8E-11	120	2.2E-10	0.35	3.9E-08	0.23
plutonium-239	0.016	1.7E-11	200	2.3E-10	0.33	3.8E-08	0.24
potassium-40	18.7	5.4E-07	0.0063	1.1E-11	6.9	7.6E-12	1,200
radium-226	15.2	6.0E-06	0.0006	1.2E-10	0.63	3.0E-09	2.3
strontium-90	13	0.0E+00[b]	–[b]	3.6E-11	2.1	6.2E-11	150
technetium-99	1.7	6.0E-13	5,600	1.3E-12	59	8.3E-12	1,100
thorium-228	0.938	5.6E-06	0.0006	5.5E-11	1.4	7.8E-08	0.12
total U	3.1	3.6E-08[c]	0.093	2.8E-11[c]	2.7	5.2E-08[c]	0.18

[a]Health Effects Summary Tables (HEAST, EPA 1992b).
[b]Not an external exposure hazard.
[c]Uranium-238 is used as a surrogate for total uranium.
ICR = Lifetime incremental cancer risk.
SF = Slope factor.
– = Not applicable.
Note: Shaded area indicates screening criterion exceeded.

Table 4-12. Preliminary Risk-Based Screening for Soil Non-Radioactive Contaminants, In Sub-Surface Infiltration Gravels/ Soils (below 15 feet), at the 200-BP-1 Operable Unit. (Sheet 1 of 3)

Parameter	Maximum Detected Soil Concentration (mg/kg)	Oral RfD (mg/kg-d)	Soil Concentration at Oral HQ=0.1 (mg/kg)	Inhalation RfD (mg/kg-d)	Soil Concentration at Inhalation HQ=0.1 (mg/kg)	Oral SF $(mg/kg\text{-}d)^{-1}$	Soil Concentration at Oral ICR = 1E-07 (mg/kg)	Inhalation SF $(mg/kg\text{-}d)^{-1}$	Soil Concentration at Inhalation ICR = 1E-07 (mg/kg)	Regulatory Soil Cleanup Guidelines (mg/kg)
Inorganics										
cadmium	3.9	1.0E-03[a]	8	–[m]	–[m]	–[n]	–[n]	6.3E+00[a]	2.6	–
chromium	119.0	5.0E-03[a]	40	–[m]	–[m]	–[n]	–[n]	4.1E+01[a]	0.4	–
copper	44	4.0E-02[d]	320	–[m]	–[m]	–[n]	–[n]	–[n]	–[n]	–
cyanide (complex)	247	2.0E-02[a,f]	160	–[m]	–[m]	–[n]	–[n]	–[n]	–[n]	–
cyanide (free)	4.1	2.0E-02[a]	160	–[m]	–[m]	–[n]	–[n]	–[n]	–[n]	–
cyanide (total)	249	2.0E-02[a,f]	160	–[m]	–[m]	–[n]	–[n]	–[n]	–[n]	–
lead	24.3	ND	–[m]	ND	–[m]	ND	–[m]	ND	–[m]	250[i], 500-1,000[a]
nickel	251	2.0E-02[a]	160	–[m]	–[m]	–[n]	–[n]	–[n]	–[n]	–
thallium	0.5	9.0E-05[a,g]	0.72	–[m]	–[m]	–[n]	–[n]	–[n]	–[n]	–
selenium	0.5	5.0E-03[a]	40	–[m]	–[m]	–[n]	–[n]	–[n]	–[n]	–
Anions										
nitrate	5,470	1.6E+00[a]	13,000	–[m]	–[m]	–[n]	–[n]	–[n]	–[n]	–
nitrite	109	1.0E-01[a]	800	–[m]	–[m]	–[n]	–[n]	–[n]	–[n]	–
Volatile Organics										
acetone	0.083	1.0E-01[a]	800	–[m]	–[m]	–[n]	–[n]	–[n]	–[n]	–
2-hexanone	0.001	5.0E-02[b]	400	9.0E-02[b,j]	150[j]	–[n]	–[n]	–[n]	–[n]	–
4-methyl 2-pentanone	0.011	5.0E-02[b]	400	2.3E-02[b]	38[j]	–[n]	–[n]	–[n]	–[n]	–

Table 4-12. Preliminary Risk-Based Screening for Soil Non-Radioactive Contaminants, In Sub-Surface Infiltration Gravels/ Soils (below 15 feet), at the 200-BP-1 Operable Unit. (Sheet 2 of 3)

Parameter	Maximum Detected Soil Concentration (mg/kg)	Oral RfD (mg/kg-d)	Soil Concentration at Oral HQ=0.1 (mg/kg)	Inhalation RfD (mg/kg-d)	Soil Concentration at Inhalation HQ=0.1 (mg/kg)	Oral SF $(mg/kg\text{-}d)^{-1}$	Soil Concentration at Oral ICR = 1E-07 (mg/kg)	Inhalation SF $(mg/kg\text{-}d)^{-1}$	Soil Concentration at Inhalation ICR = 1E-07 (mg/kg)	Regulatory Soil Cleanup Guidelines (mg/kg)
styrene	0.001	2.0E-01[a]	1,600	2.9E-01	450[f]	–[e]	–[e]	–[e]	–[e]	–
toluene	0.002	2.0E-01[a]	1,600	–[e]	–[e]	–[e]	–[e]	–[e]	–[e]	–
1,1,1-trichloroethane	0.01	9.0E-02[a]	720	3.0E-01	130[f]	–[e]	–[e]	–[e]	–[e]	–
Semi-Volatile Organics										
benzoic acid	0.710	4.0E+00[a]	32,000	–[e]	–[e]	–[e]	–[e]	–[e]	–[e]	–
bis(2-ethylhexyl)phthalate	2.7	2.0E-02[a]	160	–[e]	–[e]	1.4E-02[a]	4.6	–[e]	–[e]	–
butylbenzylphthalate	1.3	2.0E-01[a]	1,600	–[e]	–[e]	–[e]	–[e]	–[e]	–[e]	–
2-chlorophenol	0.008	5.0E-03[a]	40	–[e]	–[e]	–[e]	–[e]	–[e]	–[e]	–
diethylphthalate	0.620	8.0E-01[a]	6,400	–[e]	–[e]	–[e]	–[e]	–[e]	–[e]	–
di-n-butylphthalate	2.5	1.0E-01[a]	800	–[e]	–[e]	–[e]	–[e]	–[e]	–[e]	–
di-n-octylphthalate	0.270	2.0E-02[a]	160	–[e]	–[e]	–[e]	–[e]	–[e]	–[e]	–
isophorone	0.190	2.0E-01[a]	1,600	–[e]	–[e]	9.5E-04[a]	67.4	–[e]	–[e]	–
pentachlorophenol	0.200	3.0E-02[a]	240	–[e]	–[e]	1.2E-01[a]	0.539	–[e]	–[e]	–
tributyl phosphate	94	5.0E-03[a]	40	–[e]	–[e]	–[e]	–[e]	–[e]	–[e]	–
Pesticides										
4,4'-DDT	0.002	5.0E-04[a]	4	–[e]	–[e]	3.4E-01[a]	0.18	3.4E-01[a]	48	–
endosulfan II	0.017	5.0E-05[a]	0.4	–[e]	–[e]	–[e]	–[e]	–[e]	–[e]	–
aldrin	0.007	3.0E-05[a]	0.24	–[e]	–[e]	1.7E+01[a]	0.004	1.7E+01[a]	0.96	–

Table 4-12. Preliminary Risk-Based Screening for Soil Non-Radioactive Contaminants, In Sub-Surface Infiltration Gravels/ Soils (below 15 feet), at the 200-BP-1 Operable Unit. (Sheet 3 of 3)

Parameter	Maximum Detected Soil Concentration (mg/kg)	Oral RfD (mg/kg-d)	Soil Concentration at Oral HQ=0.1 (mg/kg)	Inhalation RfD (mg/kg-d)	Soil Concentration at Inhalation HQ=0.1 (mg/kg)	Oral SF (mg/kg-d)$^{-1}$	Soil Concentration at Oral ICR = 1E-07 (mg/kg)	Inhalation SF (mg/kg-d)$^{-1}$	Soil Concentration at Inhalation ICR = 1E-07 (mg/kg)	Regulatory Soil Cleanup Guidelines (mg/kg)
PCB	0.77	–[n]	–[n]	–[n]	–[n]	7.7E+00[e]	0.0093	–[n]	–[n]	1-25[o]

[a]Integrated Risk Information System (IRIS, EPA 1993)
[b]Health Effects Assessment Summary Tables (HEAST, EPA 1991c, 1992b)
[c]EPA 1989a
[d]Superfund Technical Support Center 1991 (STSC, EPA 1991b)
[e]40 CFR 761
[f]Based on toxicity factor for free cyanide
[g]Oral RfD based on thallium-acetate
[h]Surrogate oral RfD based on chlordane
[i]Value based on inhalation of volatiles
[j]MTCA guidelines
[k]Superfund Technical Support Center (STSC, EPA 1992c)
[l]Surrogate oral RfD and inhalation RfD based on 2-butanone
[m]No RfD or SF available to evaluate this pathway
[n]Not classified as a carcinogen or not carcinogenic by this exposure route or pathway
[o]Nickel is only carcinogenic as refinery dust
– Indicates not available
ND = Not determined
HQ = Hazard quotient
ICR = Lifetime incremental cancer risk
RfD = Chronic reference dose
SF = Slope factor
Note: Shaded areas indicate screening criterion exceeded

Table 4-13. Preliminary Risk-Based Screening for Radioactive Soil Contaminants, In Sub-Surface Infiltration Gravels/Soils (below 15 feet), at the 200-BP-1 Operable Unit.

Parameter	Maximum Soil Concentration (pCi/g)	External SF[a] (pCi-yr/g)-1	Soil Concentration at External ICR = 1E-07 (pCi/g)	Oral SF[a] (pCi)-1	Soil Concentration at Oral ICR = (pCi/g)	Inhalation SF[a] (pCi)-1	Soil Concentration at Inhalation ICR = 1E-07 (pCi/g)
antimony-125	50.4	1.2E-06	0.083	8.4E-13	98.5	1.1E-11	827
cesium-137	12,900,000	2.0E-06	0.0857	2.6E-11	2.7	1.9E-11	488
chromium-51	56.6	9.2E-08	0.857	4.3E-14	1,000	3.8E-13	30,000
cobalt-60	62	8.6E-06	0.0004	1.5E-11	5.1	1.5E-10	68
plutonium-238	203	2.8E-11	128	2.2E-10	0.35	3.9E-08	0.25
plutonium-239	0.93	1.7E-11	200	2.3E-10	0.33	3.8E-08	0.24
plutonium-239+240	5850	1.7E-11	199	2.3E-10	0.33	3.8E-08	0.24
potassium-40	16.9	5.4E-07	0.806	1.1E-11	6.9	7.6E-12	1,200
radium-226	3.7	6.0E-06	0.0006	1.2E-10	0.63	3.0E-09	3
strontium-90	14,200,000	0.0E+00[b]	–[b]	3.6E-11	2.1	6.2E-11	350
technetium-99	210	6.0E-13	5,600	1.3E-12	59	8.3E-12	1,300
thorium-228	1.17	5.6E-06	0.0006	5.5E-11	1.4	7.8E-08	0.32
total U	341	3.6E-08[c]	0.003	2.8E-11[c]	2.7	5.2E-08[c]	0.13
tritium	194.2	0.0E+00[b]	–[b]	5.4E-14	1,400	7.8E-14	120,000

[a]Health Effects Summary Tables (HEAST, EPA 1992b).
[b]Not an external exposure hazard.
[c]Uranium-238 is used as a surrogate for total uranium.
ICR = Lifetime incremental cancer risk.
SF = Slope factor.
– = Not applicable.
Note: Shaded area indicates screening criterion exceeded.

Table 4-14. Contaminants Carried Through the Risk-Based Screening Process for 200-BP-1 Operable Unit Groundwater. (Sheet 1 of 2)

Inorganic Contaminants	Maximum Concentration (µg/L)	Number of Detects	Number of Analyses[a]
antimony	120	4	418
arsenic	10.9	342	418
barium	216	281	418
beryllium	10.7	47	418
cadmium	16.7	7	418
cobalt	3.5	2	418
cyanide (complex)	766	23	23
cyanide (free)	298	18	33
cyanide (total)	2,710	86	328
chromium	56.4	247	418
copper	107	21	418
lead	84.1	17	418
manganese	295	236	418
mercury	0.42	18	418
nickel	701	75	418
selenium	53.1	97	418
silver	61.3	3	418
strontium	1,420	85	88
tin	118	2	88
thallium	2.2	1	418
vanadium	76.3	185	418
ammonia	260	1	44
chloride	79,700	165	207
fluoride	1,900	70	208
nitrate	1,970,000	201	208
nitrite	1,200	3	208
sulfate	2,930,000	185	208

Table 4-14. Contaminants Carried Through the Risk-Based Screening Process for 200-BP-1 Operable Unit Groundwater. (Sheet 2 of 2)

Inorganic Contaminants	Maximum Concentration (μg/L)	Number of Detects	Number of Analyses[a]
Organic Contaminants	**(μg/L)**		
2-butanone	6	1	41
trichloroethene	1	1	43
1-3-dichlorobenzene	3	1	43
2-chlorophenol	2	1	43
benzonic acid	1	1	43
phenol	2	2	43
butylbenzylphthalate	1	1	43
fluoranthene	2	6	43
pyrene	5	6	43
4,4'-DDT	0.09	2	43
Radionuclide Contaminants	**pCi/L**		
cobalt-60	391	45	206
potassium-40	394	13	206
plutonium-238	0.56	4	181
radium-226	262	3	196
strontium-90	1.3	3	202
technetium-99	17,000	116	158
total uranium	14 (μg/L)	206	206
tritium	220,000	135	207
radium-228	1.1	2	44

[a] Does not include analyses rejected during data validation.

Table 4-15. Preliminary Risk-Based Screening for Groundwater Contaminants at the 200-BP-1 Operable Unit.
(Sheet 1 of 3)

Parameter	Maximum Groundwater Concentration (mg/L)	Oral Rfd (mg/kg/d)	Groundwater Concentration at oral Rfd HQ = 0.1 (mg/L)	Oral SF $(mg/kg/d)^{-1}$	Groundwater Concentration at Oral SF ICR = 1E-07 (mg/L)
Inorganics					
antimony	0.120	4.0E-04[a]	0.0006	--	--
arsenic	0.011	3.0E-04[a]	.0005	1.7E+00[d]	4.8E-06
barium	0.216	7.0E-02[a]	0.11	--	--
beryllium	0.011	5.0E-03[a]	0.008	4.3E+00[a]	1.9E-06
cadmium	0.017	1.0E-03[a]	0.002	--	--
cobalt	0.004	6.0E-02[b]	0.096	--	--
cyanide (total)	2.71	2.0E-02[a,c]	0.032	--	--
cyanide (free)	0.298	2.0E-02[a]	0.032	--	--
cyanide (complex)	0.766	2.0E-02[a,c]	0.032	--	--
chromium	0.056	5.0E-03[a]	0.008	--	--
copper	0.107	4.0E-02[b]	0.064	--	--
lead	0.064	ND[b]	ND[b]	--	--
manganese	0.295	5.0E-03	0.008	--	--
mercury	0.0004	3.0E-04[b]	0.0005		
nickel	0.701	2.0E-02[a]	0.032	--	--
selenium	0.053	5.0E-03[a]	0.008	--	--
silver	0.061	5.0E-03[a]	0.008	--	--
strontium	1.4	8.8E-01[a]	1.4	--	--
tin	0.118	6.0E-01[a]	.96	--	--
thallium	0.002	9.0E-05[a]	0.0001	--	--
vanadium	0.076	7.0E-03[a]	0.011	--	--
ammonia	0.3	3.4E+01[a]	54.4	--	--
chloride	79.7	--[e]	--[e]	--	--
fluoride	1.9	6.0E-02[a]	0.096	--	--
nitrate	1,970	1.6E+00[a]	2.6	--	--
nitrite	1.2	1.0E-01[a]	0.16	--	--
sulfate	2,930	--[e]	--[e]	--	--

Table 4-15. Preliminary Risk-Based Screening for Groundwater Contaminants at the 200-BP-1 Operable Unit.
(Sheet 2 of 3)

Parameter	Maximum Groundwater Concentration (mg/L)	Oral Rfd (mg/kg/d)	Groundwater Concentration at oral Rfd HQ = 0.1 (mg/L)	Oral SF $(mg/kg/d)^{-1}$	Groundwater Concentration at Oral SF ICR = 1E-07 (mg/L)
Volatile Organics					
2-butanone	0.006	5.0E-02[a]	0.080	--	--
trichloroethene	0.001	--	--	1.1E-02[b]	0.0007
Semi-Volatile Organics					
1-3, dichlorobenzene	0.003	9E-02[a,f]	0.144	--	--
2-chlorophenol	0.002	5.0E-02[a]	0.08	--	--
benzoic acid	0.001	4.0E+0[a]	6.4	--	--
phenol	0.002	6.0E-01[a]	0.96	--	--
butylbenzylphthalate	0.001	2.0E-01[a]	0.32	--	--
flouranthene	0.002	4.0E-02[a]	0.064	--	--
pyrene	0.005	3.0E-02[a]	0.048	--	--
Pesticides					
4,4'-DDT	0.00009	5.0E-04[a]	0.0008	3.4E-01[a]	0.00002
Radionuclides	pCi/L	Oral SF $(pCi)^{-1}$	Groundwater Concentration at Oral SF ICR = 1E-07 (pCi/L)		
cobalt-60	391	1.5E-11[c]	0.300	--	--
potassium-40	394	1.1E-11[c]	0.420	--	--
plutonium-238	0.56	2.2E-10[c]	0.021	--	--
radium-226	262	1.2E-10[c]	0.038	--	--
strontium-90	1.3	3.6E-11[c]	0.130	--	--
technetium-99	17,000	1.3E-12[c]	3.500	--	--
total uranium	14(µg/L)	2.8E-11[c,e]	0.16(µg/L)	--	--
radium-228	1.1	1.0E-01[c]	0.046	--	--

Table 4-15. Preliminary Risk-Based Screening for Groundwater Contaminants at the 200-BP-1 Operable Unit.
(Sheet 3 of 3)

Parameter	Maximum Groundwater Concentration (mg/L)	Oral Rfd (mg/kg/d)	Groundwater Concentration at oral Rfd HQ = 0.1 (mg/L)	Oral SF (mg/kg/d)$^{-1}$	Groundwater Concentration at Oral SF ICR = 1E-07 (mg/L)
tritium	220000	5.4E-14[a]	85		

[a] Integrated Risk Information System (IRIS, EPA 1993)
[b] EPA Region 10
[c] Health Effects Assessment Summary Tables (HEAST, EPA 1991c or 1992b)
[d] Based on proposed arsenic unit of risk of 5E-05 ug/L (IRIS, EPA 1993)
[e] Based on toxicity facotr for free cyanide.
[f] Oral RfD for 1-2 dichlorobenzene used as surrogate.
[g] Uranium-238 used as surrogate for total uranium.
[h] Oral RfD or SF is not currently available, however the maximum value exceeds the current and proposed MCL of 0.05 and 0.015 respectively.
[i] Oral RfD or SF is not currently available, however the maximum value does not exceed the secondary MCL of 250 mg/L.
[j] An oral RfD or SF is not currently available, however the maximum value exceeds the secondary MCL of 250 mg/L.
-- = Not available or not applicable.

Table 4-16. Maximum Detected Concentrations of Radionuclides of Potential Concern in Each Crib within the Near Surface Soils (0-15 ft.).

CRIB	DEPTH m (ft)[a]	RADIONUCLIDES in pCi/g						
		Cs-137	K-40	Ra-226	Sr-90	Th-228	Total U	Mn-54
216-B-43	0.91-1.8 (3-6)	3.65		1.28		0.6		
	3.0-4.0 (10-13)		13.4		3.7			
216-B-44	0.91-1.8 (3-6)	4.89		1.28			1.5	
	2.7-3.7 (9-12)		13.4		2.3	0.71	1.5	
216-B-45	0.91-1.8 (3-6)	3.26		0.823	1.7			
	3.0-4.0 (10-13)		12.7			0.625		
216-B-46	0.91-1.8 (3-6)	0.26		0.95	0.6		1.7	
	2.7-3.7 (9-12)	0.26	13.9		0.6	0.79		
216-B-47	0.91-1.8 (3-6)	5.59	14.4	1.36		0.89	1.1	
	3.7-4.3 (12-14)				9.1		1.1	
216-B-48	0.91-1.8 (3-6)	3.63	15.6	1.55	13	0.91		
	2.7-4.0 (9-13)				13		2.5	
216-B-49	0.91-1.8 (3-6)	2.05	13.6		11	0.62		
	2.7-3.4 (9-11)			0.719	11			
216-B-50	1.2-1.8 (4-6)	4.9	13.2	1.13	0.32		1.6	
	3.4-4.6 (11-15)		13.2				1.6	
216-B-57	0.61-1.5 (2-5)	0.7	12.6	1.13	0.25		1.8	
	2.4-3.0 (8-12)		12.6			0.77		
Task 3 Soils		265	18.7	15.2	2.6	0.94	3.1	0.065
Total Number of Detects		130	123	87	109	117	57	1

[a]Depths listed are the depth range in which the maximum contaminant concentration was found for a given crib. More than one borehole and/or sample may be represented by the listed depth range.

Table 4-17. Maximum Detected Concentrations of Radionuclides of Potential Concern in each Crib within the Subsurface Infiltration Gravels/Soils (15+ ft.). (Sheet 1 of 2)

CRIB	DEPTH m (ft)*	RADIONUCLIDES in pCi/g												
		Sb-125	Co-60	Cs-137	K-40	Pu-238	Pu-239	Pu-239+240	Ra-226	Sr-90	Tc-99	Th-228	Total U	Cr-51
216-B-43	5.6-6.4 (18.5-21)		2	3440000		8.6		405		6700000			30.8	
	8.8-9.6 (29-31.5)													
	17.1-17.8 (56-58.5)	1.42			16.9				1.27			0.887		
	48.1-48.8 (158-160)		2								210			
216-B-44	6.7-7.6 (22-30)		50	2880000		55.5		626		6540000	200		95.3	
216-B-45	5.2-6.1 (17-20)			6360000		114		2350		5960000				
	6.1-6.4 (20-21)										100			
	8.2-9.1 (27-30)	23.4	2.72		15.6		0.12		1.81		100	0.734	54.5	56.6
216-B-46	4.6-5.5 (15-18)										100			
	5.8-6.7 (19-22)		2.2	364000		6.94		227		353000			35.3	
	8.2-9.1 (27-30)	50.4			14.7		0.93		2.44		120	0.516		
216-B-47	7.0-7.9 (23-26)			10200000		203		5850		14200000	50		341	
216-B-48	5.5-6.1 (18-20)			12900000						10700000	200		223	
	5.8-6.7 (19-22)		62			65.1		1200						
	8.8-9.7 (29-32)	13.4			16.5		0.022		3.61			0.853		
216-B-49	5.2-6.1 (17-20)			1640000		11		588		2080000			121	
	8.2-9.1 (27-30)						0.11		3.71					46.3

Table 4-17. Maximum Detected Concentrations of Radionuclides of Potential Concern in each Crib within the Subsurface Infiltration Gravels/Soils (15+ ft.). (Sheet 2 of 2)

CRIB	DEPTH m (ft)*	RADIONUCLIDES in pCi/g												
		Sb-125	Co-60	Cs-137	K-40	Pu-238	Pu-239	Pu-239+240	Ra-226	Sr-90	Tc-99	Th-228	Total U	Cr-51
216-B-49	14.3-14.9 (47-49)	39.9									160	1.17		
	22.9-23.8 (75-78)		4.03		16.5						160			
216-B-50	4.9-5.5 (16-18)			2000000		5.06		249		570000				
	5.8-6.4 (19-21)		28.6								132		22.6	
	8.2-8.8 (27-31)	7.6			14.7		0.24		33.8			0.609		
216-B-57	4.6-6.1 (15-20)						0.01			67			3	
	8.2-10.4 (30-34)			67000	16.6	0.01	0.1	0.003	1.01		60	0.875	3	
	17.1-17.7 (56-58)				16.6		0.1							
Total Number of Detects		10	24	30	43	37	8	38	26	58	40	43	52	2

* Depths listed are the depth range in which the maximum contaminant concentration was found for a given crib. More than one borehole and/or sample may be represented by a given depth range.

Table 4-18. Selected Radionuclide Contaminant Levels with Depth.

Crib	Depth (ft)	Strontium-90 (pCi/g)	Gross Alpha (pCi/g)	Gross Beta (pCi/g)	Borehole Temporary ID	Borehole Permanent ID
216-B-46	8-11	0.6	6.3	38	216-B-46A	299-E33-310
216-B-46	9-11	0.1	6.8	31	216-B-46B	299-E33-299
216-B-46	9-12	0.1	5.3	33	216-B-46C	299-E33-311
216-B-46	15-18	2.9	5.7	32	216-B-46A	299-E33-310
216-B-46	19-21	353,000	320	1240000	216-B-46B	299-E33-299
216-B-46	19-22	227000	138	612000	216-B-46C	299-E33-311
216-B-50	10-12	--	--	30	216-B-50A	299-E33-308
216-B-50	12-16	--	7.0	38	216-B-50B	299-E33-303
216-B-50	11-15	--	4.1	31	216-B-50C	299-E33-309
216-B-50	16-18	570000	333	2980000	216-B-50A	299-E33-308
216-B-50	16-18	67700	107.5	622300	216-B-50B	299-E33-303
216-B-50	19-21	7750	25.7	64500	216-B-45C	299-E33-309
-- = not detected in this borehole.						

Table 4-19. Contaminants Eliminated as a Contaminant of Potential Concern Based on Extent of Contamination.

Parameter	Exposure Pathway(s) at Risk	Reason
Near Surface Soil		
chromium	Inhalation	For the near surface, none of the detects were above the background concentration of 27.9 mg/kg
chrysene	Inhalation	One detect very near the instrument detection limit (IDL)
PCB	Inhalation	One detect very near the instrument detection limit (IDL)
manganese-54	External	One detect, and has a half life of 300 days
potassium-40	External, Oral ingestion	One detect, only slightly above background, naturally occurring radionuclide that is not a fission product.
Subsurface Infiltration Gravels/Soils		
chromium	Oral ingestion	Two detects above background, one at 223-228 feet which may be due to groundwater movement, and one detect at 30 feet only slightly above background.
cyanide (total) cyanide (complex)	Oral ingestion	Failed screening for only one detect and only if oral RfD for free cyanide is used. Free cyanide was tested for in the same sample and passed screening.
aldrin	Oral ingestion	One detect below CRQL.
chromium-51	External	Two detects, has a half life of 28 days and was found at 27-30 feet.
potassium-40	External, Oral ingestion	For the subsurface, none of the detects were above the background concentration of 18.6 pCi/g.

Table 4-20. Contaminants of Potential Concern and their Corresponding Upper Confidence Limits (UCLs), for Near Surface Soils and Subsurface Infiltration Gravels/Soils (~15 ft to 35 ft).

Crib(s)	Near Surface Soils (0-15 ft)				Near Surface Soils (0-15 ft)			
	216-B-43 to -50				216-B-57			
	UCL (pCi/g)	X	S	n	UCL (pCi/g)	X	S	n
cesium-137	23	14.8	41.9	71	0.49	0.326	0.196	6
radium-226	2.0	1.55	2.52	73	0.98	0.763	0.263	6
strontium-90	1.8	1.2	2.61	62	0.23	0.14	0.077	4
thorium-228	0.65	0.62	0.16	73	0.74	0.664	0.09	6
total uranium	0.77	0.64	0.65	71	1.4	0.75	0.687	5

Crib(s)	Subsurface Infiltration Gravels/Soils (15-35 ft)				Subsurface Infiltration Gravels/Soils (15-35 ft)			
	216-B-43 to -50				216-B-57			
	UCL (ug/kg)	X	S	n	UCL (ug/kg)	X	S	n
PCB	170	150	70.9	46	480	246	324	7
tributyl phosphate	16,000	8930	19600	26	–	–	–	–
	mg/kg				mg/kg			
cadmium	1.0	0.839	0.858	48	0.49	0.36	0.195	8
nickel	42	31.0	45.0	48	9.1	7.74	2.05	8
	(pCi/g)				(pCi/g)			
antimony-125	32	23.7	12.7	8	–	–	–	–
cesium-137	2,000,000	1,400,000	2,700,000	48	34,000	12,000	27,000	6
cobalt-60	42	25.6	69.7	48	–	–	–	–
radium-226	6.9	4.0	5.67	12	17	6.21	14.9	7
strontium-90	2,600,000	1,800,000	3,200,000	45	42	13.7	29.8	5
plutonium-238	32	20.4	43.8	48	0.012	0.0075	0.0079	8
plutonium-239	0.26	0.13	0.26	12	0.0063	0.0042	0.0028	7
plutonium-239+240	1,100	700	1,300	31	–*	–*	–*	–*
technetium-99	88	72.4	56.2	40	50	15.4	29.7	4
thorium-228	0.85	0.706	0.272	12	2.1	1.12	1.28	7
total uranium	64	46.3	69	43	11	4.31	10.0	8

*Only one data value for Pu-239+240
– = not a contaminant of potential concern for this soil interval.
X = Mean
S = Standard deviation
n = number of samples
UCL = upper 95% confidence limit of the mean

Table 4-21. Groundwater Sample Results for Contaminants of Potential Concern Exhibiting Limited Extent. (Sheet 1 of 4)

Parameter	Background Concentration	Screening Value Concentration	Well Number	Quarter	Sample Number	Result	Qualifier	Units
Radionuclides	pCi/L	pCi/L						
Pu-238	N/A	.021	299-E33-40	1	B00FH0	0.22		pCi/L
Pu-238			299-E34-1	3	B011B6	0.27		pCi/L
Pu-238			699-49-57a	2	B00XJ3	.56	J	pCi/L
Pu-238			699-52-54	3	B01176	0.09	J	pCi/L
Ra-226/228	N/A	0.038	299-E33-40	3	B011H7	262	J	pCi/L
Ra-226/228			299-E33-34	1	B00F56	0.78	J	pCi/L
Ra-226/228			299-E33-04	1	B00F88	.59	J	pCi/L
Ra-226/228			699-49-55b	1	B00F12	1.1	J	pCi/L
Ra-226/228			299-E33-40	1	B00FH0	0.73	J	pCi/L
Sr-90	N/A	0.13	299-E33-40	4	B019M7	1.3	J	pCi/L
Sr-90			699-50-53b	2	B00XJ6	0.59	J	pCi/L
Sr-90			299-E33-24	2	B00XY7	.35	J	pCi/L
Organics								
4,4'-DDT	N/A	0.01	699-50-53A	1	B00DZ9	0.09	J	µg/L
TCE	N/A	0.7	299-E33-12	1	B00F76	1	J	µg/L
Metals								
Ag	<10	8	699-49-55A	4	B019H8	61.3	J	µg/L
Ag			299-E33-12	3	B011K6	10		µg/L

Table 4-21. Groundwater Sample Results for Contaminants of Potential Concern Exhibiting Limited Extent. (Sheet 2 of 4)

Parameter	Background Concentration	Screening Value Concentration	Well Number	Quarter	Sample Number	Result	Qualifier	Units
As	10	0.0048	299-E33-07	5	B01R66	10.9		µg/L
As			299-E33-33	4	B019Y0	10.3		µg/L
As			299-E33-38	1	B00FP9	10.6		µg/L
Ba	68.5	110	699-55-57	4	B019D2	216		µg/L
Be	<5	8	699-55-57	4	B019D2	10.7		µg/L
Cd	<10	2	299-E33-39	4	B019M4	16.7		µg/L
Cr	<30	8	699-49-57B	2	B00XK0	56.4	J	µg/L
Cr			299-E33-35	3	B011F1	30.1		µg/L
Cu	<30	64	699-50-53A	3	B01186	107		µg/L
Mn	163.5	160	699-52-57	3	B01174	244		µg/L
Mn			699-48-50	1	B00F17	234	J	µg/L
Mn			699-52-57	2	B00XG9	225		µg/L
Mn			699-49-57B	1	B00F07	169	J	µg/L
Mn			669-52-57	1	B00DY7	295	J	µg/L
Ni	<30	32	699-50-53A	2	B00XJ1	701	J	µg/L
Ni			299-E34-05	3	B011C3	32.4	B	µg/L
Ni			699-E30-53A	1	B00F00	590		µg/L
Pb	<5	10	299-E33-14	4	B019Y8	84.1	J	µg/L
Sb	Not Available	0.6	699-55-57	4	B019D2	120		µg/L

Table 4-21. Groundwater Sample Results for Contaminants of Potential Concern Exhibiting Limited Extent. (Sheet 3 of 4)

Parameter	Background Concentration	Screening Value Concentration	Well Number	Quarter	Sample Number	Result	Qualifier	Units
Sb			299-E33-18	5	B01R75	38.4	B	µg/L
Sb			299-E33-39	2	B00XT3	29.9	B	µg/L
Se	<5	8	699-50-53A	1	B00P00	32.6		µg/L
Se			699-50-54A	2	B00XJ1	31.2	J	µg/L
Se			699-50-53A	3	B01186	25	J	µg/L
Se			699-50-53A	4	B019J2	53.1	J	µg/L
Se			699-50-53A	5	B01QZ7	38.1		µg/L
Se			699-52-54	1	B00DZ0	12.6		µg/L
Se			699-52-54	3	B01177	9.7	J	µg/L
Se			699-52-54	5	B01R09	9.9	J	µg/L
Se			699-55-57	1	B00DK7	11.3		µg/L
Se			699-55-57	2	B00XF1	11.3		µg/L
Se			699-55-57	3	B01156	13.8	J	µg/L
Se			699-55-57	4	B019D2	12.8		µg/L
Se			699-55-57	5	B01QY7	11.3	J	µg/L
Tl	Not Available	0.1	299-E33-03	2	B00XX0	2.2	B	µg/L
Anions								
fluoride	1.34	0.96	299-E33-04	1	B00F88	1.9		mg/L
fluoride			299-E33-07	1	B00FB4	1.7		mg/L

Table 4-21. Groundwater Sample Results for Contaminants of Potential Concern Exhibiting Limited Extent. (Sheet 4 of 4)

Parameter	Background Concentration	Screening Value Concentration	Well Number	Quarter	Sample Number	Result	Qualifier	Units
fluoride			699-52-57	1	B00DY6	1.7	J	mg/L
nitrite	Not Available	0.16	699-52-57	3	B01173	1.2	J	mg/L
nitrite			299-E33-01	2	B00XW0	0.25		mg/L
nitrite			299-E33-05	2	B00XZ6	0.25		mg/L
sulfate	90.5	250	699-50-53A	2	B00DJ0	2930		mg/L
sulfate			699-50-53A	1	B00DZ9	522	J	mg/L
sulfate			699-50-53A	4	B019J1	512		mg/L
sulfate			299-E33-39	2	B00XT2	448		mg/L
sulfate			699-50-53A	5	B01QZ6	431		mg/L
sulfate			299-E33-40	2	B00XT5	338		mg/L

5.0 CONTAMINANT FATE AND TRANSPORT ANALYSIS

This chapter begins with an environmental fate analysis for the contaminants of potential concern identified in Chapter 4, and concludes with an analysis of transport of these contaminants through the pathways of air, surface water, subsurface water, and biota. Although the subsurface water pathway includes both an unsaturated pathway and a saturated pathway, transport in the saturated pathway is not addressed since groundwater has been removed from this operable unit and will be addressed in the 200 East Aggregate Area Groundwater Study. Results of the fate and transport analyses presented below are used in Chapter 6 to evaluate the threats posed to human health and the environment by the contaminants of potential concern.

5.1 CONTAMINANT FATE ANALYSIS

The purpose of this analysis is to determine the behavior of each contaminant of potential concern listed in Section 4.4 in the environmental media in which it is transported. Particular emphasis is placed upon contaminant persistence and mobility. The following discussion is divided into organic, non-radioactive inorganic, and radioactive contaminants.

5.1.1 Organic Contaminants of Potential Concern

The organic contaminants of potential concern for the 200-BP-1 operable unit are PCBs and tributyl phosphate (TBP). The physical parameters for these compounds that relate to their environmental fate are presented in Table 5-1. The following information was compiled from SRC (1991b) (for PCBs) and Sax and Lewis (1987) (for tributyl phosphate).

5.1.1.1 Polychlorinated Biphenyls. Two PCB mixtures (Aroclors 1232 and 1254) were detected within soil media at the 200-BP-1 operable unit. Each Aroclor is a mixture of different congeners of chlorobiphenyl, and the relative importance of the environmental fate mechanisms generally depends on the degree of chlorination. In general, the persistence of PCBs increases with an increase in the degree of chlorination.

This group of compounds strongly adsorb to soil, and adsorption generally increases with a greater degree of chlorination and higher organic carbon content in the soil. Solubilities for PCBs are low and decrease with increasing chlorination. The higher chlorinated forms do not readily leach nor are they significantly mobile within soils under typical environmental conditions. PCBs detected in the soils at the 200-BP-1 operable unit were mainly Aroclor 1254 (three of four detections) which have a high degree of chlorination. The mobility of PCBs through the vadose zone at the 200-BP-1 operable unit is expected to be low, even in soil with low organic content typical of Hanford Site soils.

In the presence of organic solvents, PCBs may leach quite rapidly through soil. Volatilization of PCBs from soil surfaces, although slow, may be an important ultimate fate mechanism. Volatilization potential increases with decreasing chlorination.

Within natural water systems, PCBs readily adsorb to sediments and suspended matter. Although adsorption appears to immobilize the substance (especially the higher chlorinated congeners), eventual redissolution into the water phase does occur. The migration rate, particularly within ground water, can be very slow. In the absence of adsorption, PCBs volatilize relatively rapidly from water (HSDB Scientific Review Panel 1990). Strong PCB adsorption to sediment and soil, however, competes significantly with volatilization, with the higher chlorinated congeners volatilizing less readily.

Atmospheric releases of PCBs can be in the form of either vapors or fugitive dust to which the chemical is adsorbed. The dominant atmospheric transformation process is probably vapor-phase reaction with photochemically produced hydroxyl radicals. Estimated atmospheric half lives range from 4 d for monochlorobiphenyl to 83 d for pentachlorobiphenyl. Physical removal of PCBs from the atmosphere is accomplished by particulate-phase fallout or rain washout of vapor-phase and particulate-phase components.

5.1.1.2 Tributyl Phosphate. Tributyl phosphate is an odorless, colorless liquid which is soluble in water (Table 5-1). Some major applications of tributyl phosphate include solvent extraction of metal ions from solutions of fission reactor products, and use as a heat-exchange medium, hydraulic fluid, and dielectric. Tributyl phosphate is not considered a volatile compound due to its large molecular weight. Information regarding the fate of tributyl phosphate is not available.

5.1.2 Inorganic (non-radioactive) Contaminants of Potential Concern

The only non-radioactive inorganic contaminants of potential concern for the 200-BP-1 operable unit are cadmium and nickel.

5.1.2.1 Cadmium. During weathering, cadmium readily goes into solution, and in the natural environment divalent cadmium (Cd^{2+}) is the predominant form. It is also possible, however, for cadmium to form complex ions with other ions such as chloride (Cl^-), hydroxide (OH^-), bicarbonate (HCO_3^-), sulfate (SO_4^{-2}), or organic chelates (Kabata-Pendias and Pendias 1984).

Cadmium does not form volatile compounds; therefore, cadmium in the atmosphere exists as suspended particular matter. It is typically associated with very small particles (<10 µm), which are subject to long-range transport. Residence times in the atmosphere range from 1 to 10 days. Removal processes from the atmosphere are due to wet or dry deposition (Life Systems, Inc. 1991b).

Important factors that control cadmium mobility in soils are pH, cation exchange capacity, and organic matter content. Cadmium is most mobile in acidic soil (pH 4.5 to 5.5), with organic matter and sesquioxides (iron and aluminum) controlling solubility. In alkaline soils, cadmium is relatively immobile and precipitation of cadmium compounds is the likely control factor in cadmium equilibria (Kabata-Pendias and Pendias 1984).

Cadmium bioaccumulates at all levels of the food chain; however, the accumulation characteristics of cadmium (in organs rather that muscle tissue) in animals make

biomagnification unlikely (Life Systems, Inc. 1991). In general, there is a direct linear relationship between cadmium in plant material and cadmium in soil. Soluble species of cadmium are easily available to plants, but total and relative uptake of cadmium are controlled by soil pH. Although it is a nonessential element in plants, cadmium is readily absorbed by both leaf and root systems. Cadmium can accumulate in the root tissue, even when it is applied foliarly (Life Systems, Inc. 1991). Cadmium concentrations greater than 5 mg/kg (dry weight) in either soils or mature plant tissue are considered phytotoxic (Kabata-Pendias and Pendias 1984).

5.1.2.2 Nickel. During weathering, nickel readily goes into solution and then coprecipitates mainly with iron and manganese oxides. In the natural environment, divalent nickel (Ni^{2+}) is the predominant form.

Nickel does not form volatile compounds; therefore, nickel in the atmosphere exists as suspended particulate matter. Nickel is broadly distributed among aerosol size groups (SRC 1991a).

Nickel is strongly adsorbed by soil, but, to a lesser degree than lead, copper, and zinc. Important factors that control nickel mobility in soils are pH, type and amount of clay minerals, organic matter content, and the presence of iron and manganese oxides and hydroxides (SRC 1991a). Nickel sorption depends strongly on pH. In alkaline soils, sorption may be irreversible (SRC 1991a).

Although nickel is not known to be an essential element for plant growth, it is readily absorbed by plant roots from solution. Nickel uptake by plants is positively correlated with nickel concentrations in the soil. Nickel is considered phytotoxic at concentrations greater than 100 mg/kg (dry weight) in soil or greater that 10 to 100 mg/kg (dry weight) in mature plant tissue (Kabata-Pendias and Pendias 1984). Apparently, nickel is not accumulated in significant amounts by terrestrial organisms. Studies on voles and rabbits and the plants they feed on did not show any nickel accumulation (SCR 1991a).

5.1.3 Radioactive Contaminants of Potential Concern

The radioactive contaminants of potential concern for the 200-BP-1 operable unit are the following: antimony-125, cesium-137, cobalt-60, plutonium (238, 239, 240), radium-226, strontium-90, technetium-99, thorium-228, and uranium (total).

5.1.3.1 Antimony-125. Antimony-125 is a fission product with a half-life of 2.7 y. Sb-125 decays by beta emission to Te-125m, which in turn decays (by isomeric transition) to stable Te-125. As with all radionuclides, the chemical characteristics of antimony-125 (e.g., oxidation state and mobility) are independent of the radioactive properties of the element. Antimony is a metalloid element that displays both metallic and nonmetallic characteristics, and has four oxidation states: -3, 0, +3, and +5. The +3 oxidation state is the most common and stable. The two most common forms of antimony are antimony metal and antimony trioxide. Antimony metal is stable under normal conditions. It may form complex ions with organic and inorganic acids. The specification and physicochemical state of antimony are important in determining the availability of antimony for adsorption to other materials. Antimony that is incorporated in mineral lattices is considered inert and not apt to be bioavailable.

Antimony is released to the air as a particulate, or adsorbed to a particulate. Coarse particulate (>5 μm or $1.6x10^{-5}$ ft) are readily removed by gravitational settling, whereas finer particulates may take more time to be removed by wet and dry deposition. The partitioning between wet and dry deposition is dependent upon the intensity and duration of precipitation events, the form of antimony, and the particle size of the element. Antimony predominantly occurs as submicron particles that may volatilize during combustion, and condense when cooled. Antimony released into the atmosphere as an aerosol is believed to react with atmospheric oxidants to form antimony trioxide. The average atmospheric half-life has been estimated at 1.9 days for antimony metal, and 3.2 days for the more volatile antimony trioxide.

The geochemical characteristics of antimony are close to arsenic and, in part, to bismuth. The weathering actions of antimony in soils are not well known; however, it is commonly associated with iron hydroxides (Kabata-Pendias and Pendias 1984).

The binding of antimony to soils is dependent on the form of antimony and the nature of the soils. The mineral form of antimony is likely to be unavailable for binding, whereas some forms of antimony may bind to inorganic and organic ligands. Oxidation generally occurs in aerobic surface soils. Antimony can be mobile under diverse soil conditions (SRC 1990). Antimony is considered a nonessential metal, and naturally-occurring antimony has been shown to be easily taken up in plants when soluble form (Kabata-Pendias and Pendias (1984).

Data concerning the mobility of antimony in soil-plant systems are very limited. For most studies, a plant:soil concentration ratio of 0.05 can be assumed. This factor may overestimate uptake from soil by leguminous and root vegetables and garden fruits. Total uptake from soil is unlikely to exceed 0.02% of the added stable element of its radionuclides on the limited culture solution experiments that have been undertaken, root uptake can be considerable. Translocation from roots to shoots is unlikely to exceed 15% of the total plant uptake, and there may be a tendency for this translocated fraction to remain in the lower parts of the plants or be transferred to the older herbaceous tissues. The extent of translocation to development grain of fruit is not known (Coughtrey et al. 1983, 1985).

5.1.3.2 Cesium-137. Cesium-137 is a fission product with a half-life of 30 y. Cs-137 decays by beta emission to Ba-137m, which in turn decays (by isomeric transition) to stable Ba-137.

Elemental cesium is a liquid at room temperatures and could volatilize. As an ion, cesium exists only in the +1 valence state. Cesium is an alkali metal with properties similar to potassium and rubidium. In the atmosphere, most cesium becomes strongly sorbed by clay minerals and organic matter that is present as dust.

In soils, cesium is strongly sorbed by clays and organic matter which greatly restricts the downward mobility of cesium (Eisenbud 1987). Wind erosion is the most likely transport mechanism for cesium in soils.

Cesium is not an essential component of plant tissue, but plants will incorporate cesium into their tissue. Typically, the concentrations in the roots are higher than those in young leaves; thus cesium is not readily translocated throughout the plant (Kabata-Pendias and Pendias 1984).

The movement and distribution of Cs-137 in soils is influenced, to a large extent, by the type of vegetation overlying the soil. When litter horizons are present, a substantial portion of Cs-137 is bound to soil horizons and leaching, erosion, and resuspension losses are small. In soils having a poorly developed organic or litter horizon, a large proportion of added Cs-137 is strongly bound to micaceous minerals; this process will leave a comparatively small proportion of the total (15%) available for transport through the soil profile or for uptake by plants. Cs-137 bound in the top few centimeters of soil will be available for loss from resuspension and/or erosion.

Data concerning plant-soil transfer ratios for cesium isotopes are extremely variable and show a range covering four orders of magnitude. It is almost impossible to attribute a substantial portion of this variation to any major soil or plant factor. In most cases, an average soil-to-plant transfer ratio of about 0.25 can be assumed for either Cs-134 or Cs-137. The main soil factors influencing this ratio are moisture content, organic matter, clay mineral content, cation exchange capacity, pH, and soluble/exchangeable potassium contents.

Cesium/potassium relationships in both soils and plants are extremely complex. Uptake from soil depends on the stage of plant development and the species concerned as well as on the soil variables noted above. However, in most conditions, it appears that uptake from soil is limited by soil supply rather than by the potential capacity of plants to absorb cesium from soil. Notwithstanding, the removal rates of cesium from soils by most crop types rarely exceeds 1% of the total that is added to soil and is more commonly represented by a value of about 0.2% per year.

A substantial portion of applied soluble cesium can be absorbed through the surfaces of vegetation, primarily via metabolic processes linked to the development status of the species being studied. In most cases, it can be assumed that the transfer ratio for soluble cesium applied to most plant surfaces is about 3. Absorption and retention of particulate material depends on particle size and solubility. In cases of foliar deposition it appears that 5 to 30% of the total deposit can be expected to be absorbed by contaminated plants; a substantial portion of this absorbed fraction can be expected to be translocated to other parts of the plant.

5.1.3.3 Cobalt-60. Cobalt-60 is an activation product of stable Co-59. Co-60 has a half-life of 5.3 y, and decays by beta emission to stable Ni-60. Cobalt occurs in two oxidation states, +2 and +3, and the formation of the complex anion $Co(OH)_3^-$ is also possible (Kabata-Pendias and Pendias 1984).

Soil organic matter and clay content contribute to cobalt distribution and behavior. The mobility of cobalt can be enhanced by the formation of cobalt-organic chalets.

Cobalt is an essential element for plant nutrition and its uptake by plants is a function of the cobalt concentration in solution rather than total cobalt concentration in soil (Kabata-Pendias and Pendias 1984). During adsorption by plants, cobalt behaves like other heavy metals (e.g., Fe, Mn) and is transported as organic complexes.

5.1.3.4 Plutonium (238, 239, 240). Although it is naturally occurring in trace amounts, nearly all plutonium exists as the result of neutron activation of uranium in fission reactor fuel elements. Pu-238 has a half-life of 87.7 y, and decays by alpha emission to U-234.

Since U-234 is a member of the uranium series decay chain, all daughters of Pu-238 are indistinguishable from the naturally occurring daughters of U-238. Pu-239 has a half-life of 2.4×10^4 y, and decays by alpha emission to U-235, the head of the naturally occurring actinium series. Pu-240 has a half-life of 6.6×10^3 yr, and decays by alpha emission to U-236, which in turn decays to Th-232, the head of the naturally occurring thorium decay chain.

Most discharges of plutonium from nuclear facilities are composed of PuO_2, but some of the Pu may come from the evaporation of $Pu(NO_3)_4$. Unlike the tightly bound PuO_2, plutonium nitrate is readily soluble in water. Although little information is available concerning transfer of transuranic radionuclides from soils to plants, the soluble fractions of these elements in soils seem to be readily absorbed by plants (Kabata-Pendias and Pendias 1984).

5.1.3.5 Radium-226. Radium-226 is a member of the (naturally occurring) U-238 decay chain. Ra-226 has a half-life of 1622 y, and decays by alpha emission to Rn-222 (a noble gas). Radium is a member of the alkaline earth group. Oxidized elements of this group are always present in the 2+ oxidation state. Radium is chemically similar to calcium and is absorbed from the soil by plants and passes up the food chain to humans (Eisenbud 1987).

Radium-226 is frequently separated from its precursors. It is rarely found in such mass concentration as to precipitate in the presence of anions for which it has a strong affinity, particularly sulfate. Once released into waters, radium isotopes are mobile until scavenged or coprecipitated with major reactants in the water. A very common site for the accumulation of radium isotopes is in the cement-like calcium carbonate "sinter" deposited at the orifices of some hot springs (NCRP 1987).

Radium is chemically similar to calcium and is, therefore, absorbed from the soil by plants and passed up the food chain to humans. Because the radium in food originates from soil and the radium content of soil is known to be variable, there is considerable variability in the radium content of food. In addition, it is reasonable to expect that chemical factors such as the amount of exchangeable calcium in the soil will determine the rate at which radium will be absorbed by plants (Eisenbud 1987).

5.1.3.6 Strontium-90. Strontium-90 is a fission product with a half-life of 28.8 y. Sr-90 decays by beta emission to Y-90, which in turn decays by beta emission to stable Zr-90. Strontium only occurs in the +2 oxidation state.

Strontium is easily mobilized during weathering, especially in oxidizing, acidic environments. In acid environments, strontium is easily leached from the soil profile. In alkaline environments (e.g., calcareous soils) strontium may replace calcium in the formation of carbonate precipitates. Organic chalets can increase the mobility and bioavailability of strontium in the soil environment.

Strontium is not an essential element and, although strontium and calcium may compete with each other during plant uptake, strontium usually cannot replace calcium in biochemical functions (Kabata-Pendias and Pendias 1984). Generally, only 1 to 2% of strontium isotopes present in most soils can be expected to be removed by common cropping processes. In extreme conditions of acid soils and high-yielding leguminous species, this percentage may rise to 4 or 5%. The extent of strontium uptake via roots is

usually minimal when compared to foliar uptake immediately after contamination. Plant base accumulation may also be an important factor; however, there are very few data concerning either the extent or the mechanism of this process. Strontium isotopes as cations in soils are absorbed more easily by plant roots than are those present as either organic chalets or complexes. Uptake appears to occur equally by either metabolic or passive processes. In most crops, more is accumulated during the period of development of fruits. From 50 to 90% of strontium absorbed by plant roots is translocated to shoots. Root-absorbed strontium is highly mobile within the plant and accumulates in leaf veins and petioles, the endosperm of some fruit, and the peel of root crops.

5.1.3.7 Technetium-99. Technetium-99 is a fission product of U-235 with a half-life of 2.1×10^5 yr. It decays by beta emission to stable ruthenium-99. Tc-99 can exist in all valence states from +7 to -1, with the +7,+4 and 0 states being the most stable (Cataldo et al. 1989).

In aqueous and aerobic environments, such as surface soils, the pertechnetate ion (TcO_4^-) forms readily, and is the dominant Tc form. The pertechnetate ion is highly mobile in soils and sorption is directly related to the organic matter content of the soils and inversely related to soil pH. The solubility and plant availability of Tc are reduced over time, suggesting that Tc undergoes reduction (+4) and reoxidation (+7) in surface soils (Cataldo et al. 1989).

Tc-99 is only slightly retained by the absorbent soil complex. The estimated fixation of Tc in soils is usually less than 10% of the quantity supplied. Tc-99 has been shown to have greater retention in acid soils over calcareous soils. In acid soils, both the water-soluble and the organic forms of Tc are predominant, whereas water-extractable forms seem to predominate in calcareous soils (Masson et al. 1989).

Tc-99 is highly absorbed by plants. The Tc-99 present in the non-readily available fraction of plant material is rapidly released and becomes available for plant uptake (Dehut et al. 1989).

Several ions such as sulfate, selenite, molybdate, and phosphate have been shown to be competitive analogous in the absorption of TcO_4^- by plant roots; thus, the bioavailability of Tc for plant uptake may be related to its chemical similarity to some of the essential ions necessary for plant growth (Cataldo et al. 1989).

5.1.3.8 Thorium-228. Thorium-228 is a member of the naturally occurring thorium series (headed by thorium-232). Th-228 is also produced in fission reactor fuel elements. It has a half-life of 1.91 y, and decays by alpha emission to radioactive Ra-224.

Most thorium compounds common in the environment do not dissolve easily in water and do not evaporate from soil or water to the atmosphere (SRC 1989a). Thorium exists as a quadravalent ion (Th^{4+}) that readily forms hydroxy complexes ($Th(OH)_2^{2+}$, $Th_2(OH)_2^{6+}$, and $Th_3(OH)_5^{7+}$) in solutions above pH 5 (SRC 1989a).

Windblown terrestrial dust and volcanic eruptions are two important natural sources of thorium in the atmosphere in addition to anthropogenic sources. The chemical form of thorium during residence in the atmosphere may be thorium dioxide (ThO_2). Little is known of atmospheric chemical reactions; however, ThO_2 may convert to thorium sulfate ($Th(SO_4)_2$). The rate of atmospheric removal will depend on weather conditions, particle

size and density, and the chemical form of thorium particles. Residence times are likely on the order of a few days. Thorium particles (<10 μm or $3.3x10^{-5}$ ft) may travel long distances from their emission source (SRC 1989a).

With weathering, thorium is easily mobilized in the form of various complex inorganic cations and organic compounds. In soils, thorium exists as either Th^{+4} or ThO_2 and is soluble over a broad range of soil pH (Dragun 1988). Concentrations in soil generally increase with the amount of clay or organic matter present. In most circumstances thorium will remain strongly sorbed to soil and its mobility will be minimal. Groundwater leaching is possible, however, in soils with low sorption capacity or with the formation of soluble complexes or ligands. Certain microorganisms present in soil may enhance the dissolution of thorium in soils (SRC 1989a). Thorium mobility may also increase due to increased solubility in the presence of several organic acids (Kabata-Pendias and Pendias 1984).

Thorium does not readily bioconcentrate in plants (the plant/soil transfer ratio is <0.01); a ratio of 1 means concentrations in plant and soil are equal. Plants grown at the edge of impoundments of uranium tailings, however, do contain thorium and have been known to have plant/soil concentration ratios of about 3. Plant root systems adsorb thorium from the soil, but transport is not very extensive and there may be a hundred-fold difference between the root and the above ground portions (SRC 1989a).

5.1.3.9 Uranium (234, 235, and 238). Uranium is the heaviest element (atomic number 92) that is naturally occurring to any significant degree. Natural uranium is comprised of 99.28% (by weight) U-238, 0.72% U-235, and 0.0058% U-234. These ratios change as uranium is processed, such as when the element is enriched (in U-235) for use as a reactor fuel. Other long-lived isotopes of uranium (U-232, U-233, and U-236) are not naturally occurring, but are produced as the result of neutron activation of uranium in fission reactor fuel elements. Uranium-238 is the head of the uranium series decay chain, of which uranium-234 is a member. Uranium-235 is the head of the actinium series. The half-lives of uranium-238, uranium-235, and uranium-234 are $4.5x10^9$, $7.0x10^8$, and $2.4x10^5$ y, respectively. Small amounts of uranium are ubiquitously present in rocks, soil, surface water, groundwater, plants, and animals. Uranium is released to the environment by both natural and anthropogenic actions.

Particulate uranium is removed from the atmosphere by wet and dry deposition, and atmospheric transport is influenced by particle size distribution and density. Residence times are unknown, but are assumed to be on the order of that for atmospheric dust behavior (>1.0 μm or $3.3x10^{-6}$ ft). Deposition on surface water and its transport to sediments is probably the ultimate fate of atmospheric uranium (SRC 1989b).

With weathering, uranium is easily mobilized as various complex inorganic cations and in organic compounds. Ionic forms of uranium that exist in soil are: UO_2^{2+} and U^{+4} (Dragun 1988). Important reactions of uranium in soil are complexation with anions and ligands, and the reduction of U^{6+} to U^{4+}. These reactions are important in controlling the mobility in soil and water, and are influenced by redox, pH, and the sorbing characteristics of sediments and soils. In most soils, the sorption of uranium is such that it will not leach, particularly in soils containing clay and iron oxide. Maximum sorption occurs when the hydroxy complex of uranium is present. At pH >6 in the presence of high carbonate or hydroxide concentrations, uranium may form anionic complexes such as $[UO_2(OH)_4]^{-2}$. The

mobility of anionic uranium complexes in soil will be determined by the nature of the soil (SRC 1989b). Formation of hydrated uranium cations of UO_2^{+2} will result in solubility over a wide range of pH (Kabata-Pendias and Pendias 1984).

In plants, uranium uptake may be restricted to the root system and may be transported from the soil only to the outer root membrane, and not to the interior of the root. No significant translocation of uranium from soil to above-ground plant parts has been observed (SRC 1989b).

5.2 CONTAMINANT TRANSPORT ANALYSIS

The purpose of the following analysis is to provide reasonably conservative estimates of contaminant concentrations in various environmental media at points of potential receptor exposure. Many contaminant transport pathways are considered, but contaminant concentrations are calculated for only those pathways determined to be pertinent in the contaminant fate analysis. Each contaminant transport pathway consists of the following five elements (EPA 1986b):

- A contaminant source
- A contaminant release mechanism
- An environmental transport medium (or media)
- An exposure route
- A receptor.

Operable unit contaminant sources are described in Sections 3.1 and 4.1 from a physical and chemical perspective, respectively. Receptor populations are described in Section 3.6, and this information is used in conjunction with exposure routes in Chapter 6. Therefore, this section focuses on release mechanisms and environmental transport media.

5.2.1 Air Pathway

There are three possible release mechanisms by which soil contaminants can enter the atmosphere: radioactive decay, volatilization, and emission of fugitive dust. Radioactive decay is a possible release mechanism when the daughter product produced is a gas, and can apply to the contaminants, Ra-226 decaying to Rn-222 and Th-228 decaying eventually to Rn-220, from the list of potential contaminants of concern. Volatilization most often occurs when an organic liquid passes into the vapor state, and is a possible release mechanism for only one contaminant, PCB, from the list of potential contaminants of concern. Emission of fugitive dust acts as a release mechanism for contaminants which adsorb onto soil particles, and hence is applicable to all of the potential contaminants of concern.

5.2.1.1 Radioactive Decay. It was mentioned previously that radioactive decay is a potential release mechanism because Ra-226 and Th-228 have gaseous isotopes, Rn-222 and Rn-220, as their daughter products. As Ra-226 and Th-228 decay the resulting radon diffuses upward in the soil column. Radon 222 and 220 have short half lives (3.8 d and 55 s, respectively), and will decay to Po-218 and Po-216 which as solids will be contained within the soil column. Nevertheless, some radon will diffuse to land surface and into the

atmosphere. The amount of radon emissions will be minor due to the small quantity (for example the gravels/soils of cribs 216-B-43 have about 12 pCi/g of Ra-226 and 0.85 pCi/g of Th-228 compared to area specific background values of 1 pCi/g and 0.77 pCi/g, respectively) and the relatively long half-life of Ra-226 (1600 yr). For these reasons radioactive decay is not considered a major airborne release mechanism for this operable unit.

5.2.1.2 Volatilization. PCB's have such small vapor pressures and large soil adsorption constants that a rate of PCB volatilization would be extremely small. These reasons, in conjunction with the low PCB concentrations measured (UCL of 1 mg/Kg), indicate that the volatilization release mechanism is inactive.

5.2.1.3 Fugitive Dust Emission and Transport Modeling. Concentrations of fugitive dust in air and eventually its fallout to ground surface must be known to evaluate human health risks associated with inhalation, ingestion, and external exposures respectively. Transport modeling makes possible the prediction of these concentrations for anticipated future conditions. Although air and soil concentrations have been measured for current conditions, transport modeling allows distinguishing contributatory concentrations and hence contributing risks due solely to the 200-BP-1 operable unit.

Fugitive dust emission and subsequent atmospheric transport was modeled using Version 91109 of the EPA's Fugitive Dust Model (FDM; Winges 1991). The FDM is a computer code which analytically solves a commonly used air dispersion equation. Input to the FDM includes surface, soil, and meteorological information. Output from the FDM consists of airborne particulate concentration and particulate deposition mass flux at user specified locations.

5.2.1.3.1 Approach. The operable unit was divided into two source areas for emission of contaminated fugitive dust: the 216-B-57 crib, and the crib grouping of 216-B-43 through 216-B-50. Grouping the 216-B-43 through 216-B-50 cribs was justified considering the closeness of the cribs, the shared underground effluent distribution system, and the similarity in the contaminants found in the cribs. It is likely that remedial action objectives and measures would be the same for cribs 216-B-43 through -50. These same reasons were used to justify modeling fugitive dust emission from the 216-B-57 crib separately. It is possible that the 216-B-57 crib could have different remedial action objectives and remedial measures than 216-B-43 through 216-B-50 crib grouping, and as such, risks for crib 216-B-57 should be calculated separately.

Consistent with the approach used in the risk-based screening and elsewhere in this report, both source areas were divided into a near-surface soil zone and a subsurface infiltration gravel/soil zone. The subsurface infiltration gravel/soil zone consists of the materials located below 4.6 m (15 ft). The near-surface soil zone includes materials from 0-4.6 m (0-15 ft) and is comprised primarily of stabilized surface soils and crib backfill with from 0.3-1.2 m (1-4 ft) of gravel at the bottom of the zone.

A total of eight FDM simulations were run, which corresponds to two simulations for each combination of operable unit source area and soil zone. The first four simulations were conducted to calculate airborne dust concentrations from fugitive emissions. The concentrations were calculated for the operable unit sources only. It was assumed that the current protective clean soil cover (from surface stabilization activities during Task 3) is not maintained in the future and that the contaminated near surface soils are exposed and

accessible to wind erosion. For fugitive dust emissions to occur from crib infiltration gravels/soils, human intrusion must occur. These future scenarios represent a lack of clean soil cover maintenance and major disturbance of the cribs materials such as excavation but do not include removal to a secured disposal site. Fugitive dust emissions from 200-BP-1 operable unit sources are assumed to be zero for current conditions because of the clean soil cover and for future conditions if the clean soil cover is maintained.

The remaining four simulations were conducted to estimate future deposition of airborne particulates to surrounding areas. The concentrations of the 200-BP-1 operable unit contaminants on the ground resulting from fugitive dust emissions are necessary to predict human health risks from ingestion of soil, external exposure and other exposure pathways. Calculation of these concentrations requires consideration of the dust concentrations from operable unit sources as well as study area sources outside of the operable unit. This involved calculating mixing factors from the ratio of particulate deposition mass flux due solely to fugitive dust from operable unit sources to deposition mass flux due to fugitive dust from the remaining portions of the study area. Fugitive dust emissions from the study area were minimized to provide conservative mixing factors. This approach conceptualized dust deposition occurring not only from this operable unit but as a mixture from general fallout of ambient airborne particulate, much of which occurs from 600 Area fugitive dust emissions. The resulting concentrations of deposited dust are expressed as the ratio of operable unit airborne dust deposition rate over regional dust deposition rate. Application of the mixing factors in calculating ground-surface contaminant concentrations is further explained in Section 5.2.1.3.4.

5.2.1.3.2 Description. The FDM uses the Universal Soil Loss equation as simplified by Woodruff and Siddoway (1965) to calculate fugitive dust emission rates:

$$E = ACI'KL'V' \qquad 5\text{-}1$$

where:

- E = emission factor (mass/time-area),
- A = portion of total dust emission as suspended particulate matter (i.e. particle diameter less than 30 μm) (dimensionless),
- C = climatic factor ($0.0 \leq C \leq 1.0$) (dimensionless),
- I' = soil erodibility factor (mass/time-area),
- K = surface roughness factor ($0.5 \leq K \leq 1.0$) (dimensionless),
- L' = unsheltered field width factor ($0.0 \leq L' \leq 1.0$) (dimensionless), and
- V' = vegetative cover factor ($0.0 \leq V' \leq 1.0$) (dimensionless).

Wind velocity and soil moisture are major factors in windblown fugitive dust emission rates, and are reflected in the climatic factor (Cowherd et al. 1974) as:

$$C = \frac{3.863u^{w}}{(PE)^{2}} \qquad 5\text{-}2$$

where:

PE = site-specific Thornthwaite's precipitation-evaporation index = 29.1 (Weather Bureau and SCS 1962),
u = wind speed (m/s), and
w = wind speed dependance factor = 3.

Combining Equations 5-1 and 5-2 results in:

$$E = Q_0 u^w \qquad 5\text{-}3$$

where:

$$Q_0 = \frac{(3.863)AI'KL'V'}{29.1^2} = \text{proportionality constant} \qquad 5\text{-}4$$

Input to the FDM consists of the proportionality constant, Q_0, and the wind speed dependence factor, w.

The FDM models downwind air concentrations of fugitive dust with a modified Gaussian Plume formulation of airborne particulate dispersion that incorporates a gradient-transfer deposition algorithm. Standard Gaussian Plume formulations account for particulate deposition due only to the gravitational settling that occurs with particles of diameters greater than 0.030 mm (1.18×10^{-3} in). The FDM, however, has the ability to treat particulate deposition due to all mechanisms. This includes particulate deposition due to turbulent motion bringing particles of all size in contact with the ground surface.

5.2.1.3.3 Input. Presented below is a discussion of the necessary model input and its derivation. Values for the variables used to determine Q_0 in Equations 5-4, organized by source area (crib grouping 216-B-43 through 216-B-50 and crib 216-B-57), and soil zone (near surface, subsurface infiltration gravel, and Hanford Site surface) are listed in Table 5-3.

Portion of Suspended Particulate Matter (A). Values of A for soil types representative of the subsurface infiltration gravel soil zone (0.02) and the surface soil zone of the Hanford Site (0.04) were taken from *Fugitive Dust Emission Factors for the Mining Industry* (Baskett 1983). This approach was not necessary for the near-surface soil zone because hydrometer test results are available. Hydrometer tests were performed on two samples, taken at 1.7 and 2.7 m (5.5 and 9.0 ft) in borehole 216-B-61A. Values of A are 6.25% and 5.25%, respectively, which results in the average of 5.75% listed in Table 5-3.

Climatic Factor (C). Meteorological data regarding precipitation and evaporation was input into FDM via C using a Hanford Site specific PE, and hourly averaged wind speeds recorded at the meteorological station in the 200 Area during 1991.

Soil Erodibility Factor (I'). Grain size distribution plots from sieve analyses were used to determine the fraction of source area soil particles with grain size diameters exceeding 0.84 mm (0.033 in.) (Table 2). This information was used to extrapolate values for I' from Figure A-1 in *Development of Emission Factors for Fugitive Dust Sources* (Cowherd et

al. 1974). The values of I' taken from Cowherd et al. (1974) for the subsurface infiltration gravel soil zone and the Hanford Site surface soil zone were 3.45×10^{-5} g/s-m^2 (5 ton/yr-acre) and 5.18×10^{-4} g/s-m^2 (73 ton/yr-acre), respectively. For the Hanford Site surface soil zone the value of I' taken from Cowherd et al. (1974) was decreased by a factor of six to account for surface skin crusting (Skidmore and Woodruff 1968) and to provide a degree of conservatism to the mixing factors for dust deposition. With an average of 86% of its particles having diameters greater than 0.84 mm (0.033 in.), the subsurface infiltration gravel soil zone was found to have an I' of zero. The subsurface infiltration gravel soil zone is therefore not erodible, and hence not a source of fugitive dust emission. For conservatism, however, it was assumed that some of the subsurface infiltration gravel soil zone could be eroded, and a small value of I' was chosen (3.45×10^{-5} g/m^2-s or 1 t/acre-yr).

Surface Roughness Factor (K). Because the Universal Soil Loss equation (Equation 5-1) was developed for agricultural use, K was originally intended as a measure of roughness due to ridges, furrows, and large clods on unvegetated fields. Smooth unplowed fields are assigned a K of 1.0, while fields with an optimum ratio of ridge height to furrow width are given the smallest possible K value of 0.5. When applying the Universal Soil Loss equation to sparsely vegetated land such as the 600 Area, it is necessary to decrease K to account for wind resistance provided by the vegetation. For this reason the surface soil zone of the Hanford Site was assigned a K of 0.5, while the soil zones in the unvegetated and relatively smooth crib areas were assigned a K of 1.0. These values are conservative since they maximize fugitive dust emission from the operable unit source areas, minimize the emission from the study area, and hence minimize the amount of dilution of contaminants deposited on the ground surface downwind of the operable unit.

Unsheltered Field Width Factor (L'). The unsheltered field width factor, L', might better be described as a measure of the relative rate of soil erosion (Cowherd et al. 1974). It is a function of not only the width of the field in the prevailing wind direction, but also I', K, and the height of any wind barrier which may be present. A field with a zero value for L' has no potential for erosion, while a value of 1.0 indicates a maximum ability to erode. Values for L' were interpolated from the curves of Figure A-5 in Cowherd et al. (1974). The Hanford Site surface soil zone was determined to have a L' of 1.0, and the near-surface soil zone in the source areas of the 216-B-43 through 216-B-50 crib grouping and the 216-B-57 crib were determined to have L' values of 0.3 and 0.2, respectively. A small I' coupled with an approximate maximum width of the crib areas of only 100 m (330 ft) results in a negligible L' for the subsurface infiltration gravel soil zone. For conservatism, however, a small but nonnegligible value of 0.1 for L' was assigned to the subsurface infiltration gravel soil zone in both crib source areas.

Vegetative Cover Factor (V'). In its original use V' is a measure of the vegetative cover which exists during periods other than crop growing season. This cover is often crop residue standing as stubble or mulched into the soil. Since measurements of V' based on crop type is not applicable to the Hanford Site, V' was instead interpreted as the fraction of ground covered by vegetation. Based on ground cover provided by sagebrush, cheatgrass, and the like, a V' of 0.5 was assigned to the surface soil zone of the Hanford Site. A value of 1.0 for V' was assigned to the soil zones in the crib areas due to the lack of vegetation.

Meteorological Data. Meteorological data was input directly to FDM as hourly averaged: wind speed, wind direction, air temperature, atmospheric mixing height, and

atmospheric stability class. The hourly averaged information was recorded at the meteorological station in the 200 Area during 1991.

Soil Grain Size Distribution Data. A single grain size distribution was used for all three soil zones to describe the erodible fraction. The distribution chosen was an average of that obtained from sieve and hydrometer tests on two near-surface soil zone samples taken from borehole 216-B-61A from drive-tube samplers. These were the only samples taken for which both sieve and hydrometer tests were performed. It is appropriate that the same grain size distribution be used for the near-surface soil zone and the Hanford Site surface soil zone since they are comprised of the same materials. The subsurface infiltration gravel soil zone probably does not, however, contain as large a proportion of fines as the near-surface soil zone. As such, simulations involving fugitive dust emission from the subsurface infiltration gravel soil zone may slightly over predict the amount of airborne particulate, and slightly under predict the amount of particulate deposited on the ground surface.

5.2.1.3.4 Results. The FDM predicted airborne particulate concentration isopleths for fugitive dust emission from the operable unit source areas are shown in Figures 5-1 through 5-4. It is important to note that Figures 5-1 through 5-4 depict dust and not contaminant concentrations. Decay factors for projecting radionuclide UCL concentrations to the year 2018 (the assumed year in which land uses at Hanford could change) are listed in Table 5-4. Radioactive contaminant concentrations beyond 2018 would obviously be lower due to decay. These decay factors were used to calculate the operable unit source area UCL soil concentrations listed for the year 2018 and are in Table 5-5. The concentrations of airborne contaminants listed in Table 5-5 were calculated by multiplying the contaminant's projected future UCL soil concentrations by the airborne particulate concentrations. It is important to note that contaminants are typically more concentrated in the smaller size fractions of a soil media. Information on contaminant distribution with grain size does not exist for the source area soils, therefore no adjustment could be made for contamination of smaller soil particles that are subject to fugitive dust emissions. Although this method does not conservatively estimate concentrations, the modeling effort is considered conservative because the source is assumed infinite and ignores any reduction in emission factor by the potential formation of a stable coarse sand and gravel surface skin over the source soils.

Figures 5-5 through 5-8 present the results of fugitive dust deposition from the 200-BP-1 operable unit sources to surrounding land surfaces. As mentioned in Section 5.2.1.3.1, deposition of fugitive dust is expressed as isopleths of mixing factors on these figures. Mixing factors at given locations represent the mass flux deposition rate of fugitive dust from the 200-BP-1 operable unit divided by the total mass flux deposition rate of fugitive dust from the entire study area. The purpose of this modeling effort was to determine the concentrations of contaminants originating from the 200-BP-1 operable unit sources to surrounding land surfaces. As such, it was assumed that the mass flux deposition rate from the portions of the study area outside of the operable unit did not contribute contaminants. If other source operable units have an unacceptable risk from fugitive dust emissions and subsequent deposition, then the RI/FS process for that respective operable unit will need to address the issue as a remedial action objective.

Hanford Site ambient air monitoring data can be used as a check for the study area fugitive dust emissions. Ambient air quality particulate concentrations from modeling

efforts for the study area was approximately 2 ug/m^3. Air quality monitoring at a primarily upwind 600 Area station (map location #10 on Figure 3-1 in Jaquish and Mitchell 1988) located west of the 200 West Area has measured ambient particulate values averaging 33 ug/m^3 in 1987 (Jaquish and Mitchell 1988). Underestimation of study area ambient fugitive dust concentrations provided the anticipated conservatism since the mass deposition flux from study area particulate will be lower and the mixing factor will be larger (more relative mass flux deposition of 200-BP-1 operable unit contaminants).

The resulting concentration of 200-BP-1 operable unit contaminants to the land surface are calculated by multiplying the mixing factor associated with a location by the concentration UCLs of the contaminants of potential concern (accounting for decay in the future) and are presented in Table 5-6 for the year 2018. Again the year 2018 represents the earliest year in which alternative land uses (i.e., residential, agricultural, etc.) were assumed to occur at the Hanford Site and radioactive contaminant concentrations beyond 2018 would be lower. The results consist of the concentrations of contaminants in the thin veneer of dust which is deposited. These results assume that the 200-BP-1 operable unit source of contaminants are infinite and that fugitive dust emissions do not decline with time from the potential formation of a stable and protective coarse sand and gravel skin over source soils.

5.2.2 Surface Water Pathways

The potential surface water pathways for 200-BP-1 operable unit contaminant transport are:

- Run-off from storm events.
- Deposition of fugitive dust on nearby surface water bodies with subsequent transport.

The 200 Area plateau is not subject to flooding by the Columbia River, even in the event of a 50% breach in the Grand Coulee Dam (Skaggs and Walters 1981). The transport of contaminant within aquifers to surface waters are not included in this operable unit but will be assessed within the 200 Aggregate Area Groundwater Study.

Stormwater runoff from the operable unit is considered a minor pathway for this operable unit. The operable unit does not have surface water. The surface soils are permeable and meteoric precipitation rapidly infiltrates. Surface ponding could potentially occur with a given storm intensity but transport is expected to be limited locally since the duration of such intensities are predicted to be short. In addition, the most significant concentration of contaminants are buried at depths of fifteen feet and greater which are not subject to stormwater erosion and transport.

Fugitive dust deposition on West Lake or the Columbia River, the two nearest surface water bodies (B-pond is an effluent waste management system), is considered insignificant for this operable unit. Results of the airborne fugitive dust transport modeling and subsequent deposition of dust in Section 5.2.1.3.4 indicate that transport of fugitive dust from the operable unit does not occur in the direction of West Lake, and that

deposition of operable unit dust is insignificant at locations as far away as the Columbia River.

5.2.3 Subsurface Water Pathways

Pore space in soil is typically filled with water, air, or a combination of water and air. When pores are filled only with water the soil is considered saturated, and when pores contain air or air and water the soil is considered unsaturated. The unsaturated or vadose zone consists of all soil between the ground surface and the water table. The saturated zone lies beneath the water table of unconfined aquifers, or the confining layer of confined aquifers. The physics describing movement of water in the saturated and unsaturated zones are different, so the two zones are frequently separated into different pathways.

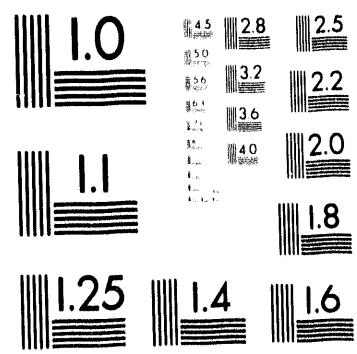
By definition, this operable unit does not include the saturated pathway. A portion of the saturated zone was, however, incorporated into the unsaturated pathway. The unsaturated pathway modeled in Section 5.2.3.2 lies between the bottom of the cribs and the bottom of the unconfined aquifer directly beneath the cribs. Inclusion of the saturated zone in the unsaturated pathway was necessary to determine which of the potential contaminants of concern reach the water table at unacceptable concentrations. Although the 200-BP-1 operable unit is a source operable unit, the prediction of future potential impacts to the underlying unconfined aquifer is needed to formulate remedial action objectives and evaluate appropriate source control measures.

5.2.3.1 Saturated Pathway Modeling. Migration of contaminants in the unconfined and confined aquifers beneath the operable unit will be investigated and modeled as part of the 200 East Aggregate Groundwater Study conducted by the Geoscience group of Westinghouse Hanford.

5.2.3.2 Unsaturated Pathway Modeling. Modeling moisture conditions and contaminant migration within the vadose zone was necessary for two reasons: to (1) identify those contaminants which may potentially traverse the unsaturated zone and enter the underlying unconfined aquifer; and (2) to provide estimates of the maximum lateral extent of liquid effluent migration. Source control remedial action objectives will be determined in part by the potential future impacts to the unconfined aquifer as predicted by the unsaturated pathway modeling. Similarly, the subsurface lateral extent of soil contamination may be required to evaluate source control measures in the feasibility study.

Migration of the liquid effluent laterally and between the cribs and the water table was numerically simulated using Version 1.2 of PORFLO-3. PORFLO-3 is an approved computer code for simulating contaminant transport in variably saturated media at the Hanford Site (DOE-RL 1991). Crib 216-B-57 was not included in the modeling scenarios described below because it received only low-level liquid effluent and concentrations are lower than within cribs 216-B-43 through 50. Conclusions resulting from the modeling effort for cribs 216-B-43 through 50 should be applicable for crib 216-B-57 for common contaminants having similar concentrations.

Selection of the contaminants to model occurred prior to the identification of the preliminary contaminants of concern discussed in Section 5.1, and hence not all of the


1.0
1.1
1.25
1.4
1.6
1.8
2.0
2.2
2.5
2.8
3.2
3.6
4.0
4.5
5.0
5.6

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potential contaminants of concern were modeled. The following is a list of the contaminants of potential concern which were modeled:

- TBP
- Cs-137
- Co-60
- Nitrate
- Sr-90
- Tc-99
- Pu-238
- Pu-239
- Pu-239/240
- total U.

For simplicity, all three plutonium species were lumped together and assumed to have the half-life of the longest lived of the three (Pu-238, 4.51×10^9 yr). Nitrate was not determined to be a contaminant of potential concern for source soils in Chapter 4.0 because its toxicity in soil is low. Nitrate has been disposed within the 200-BP-1 cribs (Appendix H) in large quantities. A nitrate groundwater plume is also present in the unconfined aquifer beneath and downgradient (north) of the 200-BP-1 operable unit. Therefore, it was modeled to predict transport and to better understand the potential future groundwater impacts.

The stable daughter products that would eventually occur from radioactive decay were not modeled, although some are inherently toxic. Only small amounts of daughter products occur from uranium, technetium, and plutonium decay because their half-lives are long. The radioactive contaminants of potential concern that have relatively short half-lives and have significant mass in the crib soils include Co-60, Cs-137, and Sr-90. Complete decay of these isotopes from the source 95% UCLs to the stable daughter results in the following source concentrations:

- Co-60 (42 pCi/g) → Nickel (3.7×10^{-8} mg/kg)
- Cs-137 (2.0×10^6 pCi/g) → Barium (2.3×10^{-2} mg/kg)
- Sr-90 (2.6×10^6 pCi/g) → Yttrium (1.9×10^{-2} mg/kg)

Since complete conversion from the parents to the stable daughters results in such low concentrations, these stable daughters were not modeled for their potential impact on the underlying aquifer.

The following is a list of contaminants of potential concern which were not modeled:

- Cadmium
- PCB
- Ra-226
- Nickel
- Sb-125
- Th-228.

These contaminants were not modeled because Ra-226, cadmium and nickel are present at low levels, PCB's are very immobile, and Sb-125 and Th-228 are present at low levels and have short half-lives.

5.2.3.2.1 Approach. The following three scenarios were modeled using PORFLO-3:

A) Vertical migration of contaminants from a single crib source with a unrealistically high natural infiltration rate (22.52 cm/yr)

B) Vertical migration of contaminants from a single crib source with a more realistic natural infiltration rate (1 cm/yr)

C) Vertical migration of contaminants from a single crib source with an infiltration rate of 10 cm/yr

D) Potential lateral migration of liquid effluent from cribs 216-B-43 through 216-B-50.

Symmetry necessitated modeling only one-quarter of the single crib used in Scenarios A, B and C.

Scenario A. The purpose of Scenario A was to identify those contaminants which may potentially traverse the vadose zone between the crib base and the water table and enter the underlying unconfined aquifer assuming extreme recharge rates (22.54 cm/yr, see Section 5.2.3.2.5) from meteoric infiltration. This recharge rate is considered extreme because it exceeds the long term average area precipitation of about 16 cm/yr (Section 3.2.1). Scenario A maximized vertical contaminant migration to screen contaminants that do not have the potential to impact the unconfined aquifer. Contaminants not entering the unconfined aquifer in Scenario A were those which either quickly decay or readily adsorb to soil particles and become bound up in the upper soil horizon. Those contaminants that entered the unconfined aquifer under the conditions of Scenario A were then modeled in Scenario B. It was not necessary to model Tc-99 in Scenario A because it appears to have already traveled the thickness of the vadose zone and entered the groundwater directly beneath the 200-BP-1 operable unit. Technetium-99 was, however, modeled in Scenario B.

Scenario B. Modeling of vadose zone migration in Scenario B used more realistic estimates of recharge rates (1 cm/yr) from meteoric infiltration than those used in Scenario A. Only those contaminants not screened in Scenario A were modeled, and results provided estimates for:

- Contaminant distribution within the unsaturated zone with time,
- Contaminant concentration within the unconfined aquifer directly beneath the crib with time.

Scenario C. Scenario C modeled vadose zone migration of contaminants using a meteoric infiltration/recharge of 10 cm/yr. This recharge rate is high (represents about 60 percent of total precipitation). Scenario C simulation was only run for contaminants (uranium) that did reach the water table within Scenario A but did not reach the water table within Scenario B at unacceptable concentration in the foreseeable future (<1000 years). Scenario C provides information on the sensitivity of infiltration/recharge rates on the transport and potential future impacts to the unconfined aquifer for uranium.

Scenario D. Scenario D provided estimates of the lateral extent of liquid effluent migration with time. Because of adsorption and radioactive decay were ignored, these estimates of lateral spreading are overly conservative for most of the potential contaminants of concern.

5.2.3.2.2 Modeled Domain. For Scenarios A, B and C the modeled domain was a single crib, whereas for Scenario D, the domain modeled was that surrounding the crib

grouping of 216-B-43 through 216-B-50. These cribs form two rows (216-B-43 through 216-B-46 and 216-B-47 through 216-B-50) in the north-south direction. The distance between the surface of the cribs is approximately 30 m (100 ft) in the east-west direction, and 90 m (300 ft) in the north-south direction. The cribs are identical in size and shape, roughly 23 m (75 ft) square at surface, tapering to 9 m (30 ft) square at base.

Each crib was filled with gravel to a depth of about 2.4 m (8 ft), and then covered with a membrane barrier and a minimum of 2.1 m (7 ft) of earth backfill. Liquid effluent was released near the bottom of the cribs in a zone of 7.6 cm (3 in) minimum diameter gravel. For the purposes of modeling, the liquid effluent was assumed to infiltrate only along the bottom surface of the cribs.

The vertical dimension of the model extends from the bottom of the cribs to the bottom of the unconfined aquifer. The thickness of the unsaturated zone between the cribs and the water table is approximately 59 m (194 ft), and beneath the cribs the thickness of the unconfined aquifer is approximately 3 m (11 ft).

5.2.3.2.3 Stratigraphy. The same conceptual model of stratigraphy was used for Scenarios A through C. The modeled stratigraphy consists of four flat-lying soil layers as described by Hoffmann (1992) and Hoffmann et al. (1992). The uppermost layer is a sandy gravel 6 m (20 ft) in thickness. Directly beneath the sandy gravel is a 46 m (150 ft) thick sand layer and a 1 m (3 ft) thick silt layer. The bottommost layer is a silty sandy gravel 10 m (32 ft) thick, which was divided into an unsaturated zone and a saturated zone so that the unconfined aquifer could be modeled. Directly beneath the bottommost layer is the relatively impermeable Elephant Mountain Member of the Saddle Mountain Basalt.

5.2.3.2.4 Finite Difference Mesh. The modeled domain was discretized to create a mesh consisting of nodes, cells, and associated cell faces. Cell faces lie midway between each pair of nodes, and typically represent physical boundaries such as the interface between soil layers. A ground surface cell face hence has one grid node a specified distance above the ground surface (in the air) and another grid node equidistant below the ground surface.

Scenario A. For Scenario A the mesh was defined by 10 nodes in each direction in the horizontal plane, and 33 nodes in the vertical plane (Figure 5-9). A square of four cells along the top surface of the mesh represents the bottom of the crib portion modeled. For reasons of numerical stability, the density of nodes was made greater at soil layer interfaces and near the bottom of the cribs.

Scenario B. The same finite difference mesh was used for both Scenario A and Scenario B.

Scenario C. The same finite difference mesh was used for Scenario C as used for both Scenarios A and B.

Scenario D. For Scenario D, the mesh was defined by 23 nodes in the horizontal plane west to east, 33 nodes in the horizontal plane south to north, and 33 nodes in the vertical plane (Figure 5-10). Symmetry could not be used to simplify this scenario because the boundary conditions varied along the top surface of the mesh. For reasons of

numerical stability the density of nodes was made greater at soil layer interfaces and near the bottom of the cribs.

5.2.3.2.5 Boundary Conditions. All boundaries of the modeled domain were assigned either a flux or a pressure for solving the equation of flow for Scenarios A through C. Scenarios A and B also required boundary values of mass flux or concentration for solving the equation of contaminant transport.

Scenario A. For Scenario A, the flow equation boundary conditions were applied to the bottom and sides of the mesh as follows:

- Zero flux along the bottom boundary
- Zero flux along the side boundaries in the unsaturated zone
- Constant head along the north and east side boundaries in the saturated zone
- Zero flux along the south and west side boundaries in the saturated zone to account for the use of symmetry.

Boundary conditions along the top of the mesh were defined as constant flux whose value outside of the crib area equaled an assumed maximum recharge rate due to meteoric infiltration. For periods when the crib was not receiving liquid effluent the entire top of the mesh was prescribed a constant flux equal to the recharge rate from meteoric infiltration, but when the crib was operating the recharge rate from meteoric infiltration was superimposed on the liquid effluent discharge rate. The liquid effluent discharge rate chosen was that corresponding to crib 216-B-45. The reason for choosing 216-B-45 was that the total liquid effluent discharged to the crib was about 4.9×10^6 L (1.3×10^6 gal), which is a middle of the range value for total volumes discharged to the 216-B-43 through 216-B-49 cribs (2.1×10^6 to 6.7×10^6 L or 5.5×10^5 to 1.8×10^6 gal), and that a high inventory of Sr-90 and Cs-137 was disposed to the crib.

The recharge rate from meteoric infiltration for Scenario A was based on the results from the T106 tank leak study conducted by Smoot et al. (1989). In that study unsaturated flow in a 2.0 m (6.6 ft) thick layer of proxy sandy gravel was numerically simulated using the computer program UNSAT-H with a combination of recorded and simulated climate data. The UNSAT-H model predicted recharge below a depth of 2.0 m (6.6 ft) equal to about 77% of precipitation or roughly 13 cm/yr (5 in/yr). The authors provided a table of results containing annual recharge rates beginning in 1947 and extending to 2021. These recharge rates were used for this scenario until the year 1992, when a constant rate equal to the maximum from the study (22.52 cm/yr or 8.87 in/yr) was applied for the remainder of the simulations.

Contaminant transport equation boundary conditions for Scenario A were applied to the bottom and sides of the mesh surfaces as follows:

- Zero diffusive flux along the bottom boundary
- Zero diffusive flux along the side boundary of the unsaturated zone

- Zero concentration along the north and east side boundaries of the saturated zone

- Zero diffusive flux along the south and west side boundaries of the saturated zone to account for the use of symmetry.

Boundary conditions along the top of the mesh outside of the crib area were prescribed as zero contaminant concentration. Within the crib area the boundary conditions were assigned zero concentration for non-operational periods, or constant nonzero concentration for operational periods.

Radionuclide concentrations for boundary conditions within the crib area were calculated by first picking the maximum inventory, if available, of each radionuclide regardless of crib from Appendix H (DOE-RL 1992a). These inventories were converted to mass by dividing by the specific activity for each radionuclide (DHEW 1970). This mass was then converted to a concentration by dividing the mass by the total volume of liquid effluent discharged to crib 216-B-45 (Table 5-7). Hence, the concentration of each radionuclide was calculated by combining the maximum inventory measured in any one of the cribs with the total liquid effluent discharge volume to crib 216-B-45, which represents a median total volume discharge for cribs 216-B-43 through 216-B-49. No inventory was available for TBP and Tc-99, so a unit concentration was assumed for source discharges. Table 5-8 summarizes the source inventory and effluent concentrations used in the vadose zone model.

Scenario B. Boundary conditions for Scenario B differed from those of Scenario A only in the recharge rate from meteoric infiltration. The large precipitation recharge rates of Scenario A were appropriate because the purpose of the scenario was to provide overly conservative estimates of vertical contaminant migration. Scenario B, however, was to produce more realistic estimates of contaminant movement, so a recharge rate from meteoric infiltration was chosen that is more representative of the 200 Area plateau.

Although the true recharge rate from meteoric infiltration is unknown, most literature suggests a rate of less than 1 cm/yr (0.4 in/yr) for the 200 Area plateau. Gee and Heller (1985) report that other authors suggest a recharge rate of 0.3 cm/yr to 1 cm/yr (0.1 to 0.4 in/yr) depending on unsaturated hydraulic conductivity, depth, and corresponding matric potential gradient. Rockhold et al. (1990) in a report of the 18.5 m (60.7 ft) deep lysimeters at the 200 East Area Lysimeter Site stated that "numerous reports (e.g., Isaacson et. al. 1974; Last et. al. 1976; Routsen et al. 1988) cite data from this facility as evidence for zero, or nearly zero, recharge.", and "that no measurable recharge has occurred during the last 18 years." Bierschenk (1959) reported "Rainfall at most points on the project [Hanford Works] has been observed to infiltrate to a depth of only a few inches or a few feet below the land surface with few exceptions. There is therefore believed to be essentially no periodic wetting of the soil at depth with rain water."

The above observations generally refer to undisturbed areas with natural vegetation. The soil overlying the cribs is neither undisturbed or vegetated, but is, however, native and separated from the cribs by an impermeable membrane. This membrane likely slows infiltrating precipitation so that evaporation has longer to occur. Whether the membrane induced evaporation is equivalent to plant transpiration from natural vegetation is

unknown, so the largest value (1 cm/yr or 0.4 in/yr) from the range of recharge rates from meteoric infiltration cited in the aforementioned reports was chosen for Scenario B.

Scenario C. Boundary conditions of Scenario C differed from those of Scenarios A and B only in the recharge rate from meteoric infiltration. A recharge rate of 10 cm/yr was used to better understand the sensitivity of contaminant transport to assumed recharge rates.

Scenario D. The boundary conditions for Scenario D are identical to those of the flow equation for Scenario B with just two exceptions: Scenario B is symmetric while Scenario D is not, and Scenario B includes infiltration from just one crib while Scenario D includes infiltration from all eight (216-B-43 through 216-B-50). The lack of symmetry required the south and west sides of the saturated zone in Scenario D to be treated as constant head. Infiltration from the additional cribs was simulated by assigning a liquid effluent discharge rate to each crib during the simulated time period when the crib operated. The liquid effluent discharge rates and the periods of discharge are listed in Table 5-7.

5.2.3.2.6 Initial Conditions. Pressures representative of conditions just prior to crib operation were assigned to the entire modeled domain for solving the equation of flow for Scenarios A through D. Scenarios A, B and C also required initial contaminant concentrations for solving the equation of contaminant transport.

Scenario A. The approximate elevation of the water table was used as the preliminary initial pressure in the saturated zone, with a small ($6x10^{-5}$) gradient in the northerly direction. The Hanford environmental impact statement for disposal of tank wastes (DOE 1987) contains varying results for groundwater movement beneath the operable unit depending on the areal distribution of the assumed natural recharge rate. For a 0.5 cm/yr (0.2 in/yr) recharge rate in the crib area there is essentially no gradient, while for a 5 cm/yr (2.0 in/yr) rate there is a slight northerly gradient comparable to that assumed for this scenario. As such a recharge rate of 5 cm/yr (2.0 in/yr) was applied to the surface of the modeled domain to reach the steady-state conditions described below.

Initial pressures in the unsaturated zone were determined by providing PORFLO-3 with the best available preliminary values, and then running PORFLO-3 until steady state was reached. The preliminary initial pressure for each soil layer was determined from the unsaturated hydraulic conductivity and moisture retention curves for that layer, and by assuming a unit hydraulic gradient. The recharge rate from meteoric infiltration was used as the unsaturated hydraulic conductivity to determine preliminary moisture contents from the unsaturated hydraulic conductivity curve. The moisture content was then used with the moisture retention curve to determine the preliminary initial pressure.

Initial contaminant concentrations in both the saturated and unsaturated zones were assumed zero for the entire modeled domain.

Scenario B. Procedures for establishing initial conditions for Scenario B were identical to those used for Scenario A, except that the initial moisture contents and pressure heads reflected the lower recharge rate from meteoric infiltration (1.0 cm/yr).

Scenario C. Procedures for establishing initial conditions for Scenario C were identical to those used for Scenario A.

Scenario D. The same initial pressures were used for Scenario D as for Scenario B, with the exception that the gradient in the saturated zone was increased for Scenario D. The need for this increase became evident when the PORFLO-3 predicted steady-state solution demonstrated saturated zone flow out of both the north and south boundaries. The new gradient was determined by running PORFLO-3 to steady state without allowing flow through the west and east boundaries of the saturated zone. The initial pressures in the saturated zone along the west and east boundaries were then set to the pressures obtained at steady state.

5.2.3.2.7 Physical Properties. Information on the following physical properties were required for input to PORFLO-3:

- Saturated hydraulic conductivity
- Unsaturated hydraulic conductivity
- Porosity
- Specific storativity
- Dry bulk density
- Distribution coefficient
- Molecular diffusivity
- Dispersivity
- Decay.

Values used for the above physical properties are identical for all four scenarios with the exception that those properties related to contaminant transport (partitioning coefficient, molecular diffusivity, dispersivity, and radionuclide half-life) were not required for Scenario D. Table 5-9 lists the van Genuchten parameters (discussed below) for calculating the unsaturated hydraulic conductivities, saturated hydraulic conductivities, porosities, specific storativities, and dry bulk densities for all four soil layers. The distribution coefficients, molecular diffusivities, and radionuclide half-lives are listed in Table 5-10 for all contaminants modeled.

Saturated Hydraulic Conductivity. For Scenarios A, B, and C, the laboratory measured saturated hydraulic conductivity was assumed isotropic. This assumption minimized the amount of lateral moisture migration, and hence exaggerated vertical moisture movement. For Scenario D, the saturated horizontal hydraulic conductivity was assumed to be five times larger than the vertical. This anisotropic ratio is in the two to ten times range typical of sedimentary deposits.

Unsaturated Hydraulic Conductivity. Unsaturated hydraulic conductivities for all three scenarios were calculated by PORFLO-3 using the van Genuchten parameters α and n. These parameters were generated by fitting a curve to moisture retention information relating matric potential to moisture content determined from laboratory testing of field samples. Results from testing of the field samples were separated into four imaginary bins on the basis of grain size distribution. Each bin corresponds to one of the four soil types described in Section 5.2.3.2.3. The sample with the highest saturated hydraulic conductivity was selected from each bin for generating the van Genuchten parameters using the RETC

computer program (van Genuchten et al. 1991). The van Genuchten curves for the four soil types are presented in Figures 5-11 through 5-14.

Porosity. The same effective, total, and diffusive porosities were used for all three scenarios. The effective porosity was taken as the saturated moisture content calculated by the RETC computer program. The total and diffusive porosities were assumed identical, and calculated by adding the effective porosity to the residual moisture content generated by RETC.

Specific Storativity. The same specific storativities were used for all three scenarios, and were calculated by rounding the effective porosities up to the nearest tenth.

Dry Bulk Density. The same laboratory measured dry bulk densities were used for all three scenarios.

Distribution Coefficient (K_d) Values for the distribution or partition coefficients were taken from literature, and were the same for Scenarios A, B, and C. Those values presented in Table 5-10 which were taken from Ames and Serne (1991) and Cantrell and Serne (1992) were derived from tests of Hanford Site soils. The Ames and Serne (1991) results are for low dissolved solids, low organic content, and neutral to basic pH conditions. The Cantrell and Serne (1992) results are for Hanford Site ambient conditions. Nitrate was assumed to be very mobile with a K_d of 0.

The range of values from two different sources are presented in Table 5-10 for Co-60. Although the Strenge and Peterson (1989) range was calculated with the same neutral to basic pH conditions as that of Cantrell and Serne (1992), the former range is probably more representative of the highly mobile Co-60 found in the operable unit. Apparently the mobility is a result of Co-60 complexing with other chemicals in the liquid effluent. Because results calculated using even the smallest Strenge and Peterson (1989) distribution coefficient (1.94 mL/g or 0.23 gal/lb) appeared too conservative compared to field observations, additional simulations were conducted using an assumed distribution coefficient of 0.5 ml/g (0.06 gal/lb).

Molecular Diffusivity. An assumed molecular diffusivity of 1 cm^2/d (0.2 in^2/d) taken from DOE (1987) was used in both Scenarios A and B for all contaminants and soil layers. This compares with the 1.7 cm^2/d (0.26 in^2/d) presented in Freeze and Cherry (1979). In either case the contribution of molecular diffusion to contaminant migration is very small for advection dominated transport.

Dispersivity. Values for the longitudinal and transverse dispersivities used in Scenarios A and B were assumed equal to 1.0 and 0.1 m (3.3 and 0.3 ft), respectively, for all four soil layers. These values were determined from previous site performance analyses at the Hanford Site, and generally agree with the results of 1.32 and 0.13 m (4.33 and 0.43 ft) presented in Gelhar et al. (1985).

Decay. Radioactive decay rates for all radionuclides modeled in Scenarios A and B are presented in Table 5-10 as half-lives. TBP and nitrate were assumed to be persistent and not degrade appreciably with time. This assumption is conservative for TBP which is an organic compound as is expected to degrade somewhat with time.

5.2.3.2.8 Modeling Results

Special Interpretation of PORFLO-3 Saturation Results. Moisture retention tests were conducted only on soil material that passed through the 2 mm (0.08 in.) openings of the #10 sieve. Soil material of diameter greater than 2 mm (0.08 in) is considered coarse sand and gravel, and typically contains very little moisture. As a result, if the van Genuchten parameters input to PORFLO-3 were calculated from a sample which was one-half gravel, the saturation values calculated by PORFLO-3 will be two-times too large. It is best to consider relative differences in PORFLO-3 predicted saturations, rather than absolute saturation values.

Comparison of Measured Versus Predicted Present-Day Soil Concentrations. Comparing measured versus predicted soil and vadose water concentrations is the best method available to assess the accuracy of the modeling results presented above. As such the Sr-90 and Cs-137 results of Scenario A, and the nitrate, Co-60, and total U results of Scenario B were compared to the recently measured soil concentrations and measured leachate. Soil concentrations are provided in Chapter 4, and summarized by crib area and soil zone in Table 4-20. The only the portion of Table 4-20 related to the crib grouping 216-B-43 through 216-B-50 and the subsurface infiltration gravel/soil zone is appropriate for comparison since the domain modeled is the area beneath the base of cribs 216-B-45 through 216-B-50. The soil concentrations predicted in Scenarios A and B compared favorably to those recently measured (Table 5-11) (please note that concentrations measured in the whole soil samples represent both water phase and solid phase contamination).

Leachate concentrations were measured during the column leach test (Task 10) and are provided in Gillespie (1992). The maximum leachate concentrations are also presented in Table 5-11 and compared favorably to model-predicted maximum vadose pore water concentrations. The only exception is nitrate where the model is predicting significantly higher concentrations (≈50,000 mg/l) compared to the maximum concentration of nitrate in column leach samples (2000 mg/l). Nitrite being a very mobile constituent would be expected to migrate with the discharge effluent. The high model-predicted nitrate concentrations indicate significant amounts of discharged effluent remains in the vadose zone. In actuality, most of the original effluent may have entered the water table as evidenced from the shape of groundwater plumes of mobile contaminants in the study area (Appendix K). The model may not be allowing transient moisture to drain as rapidly as may be actually occurring.

Scenario A Contaminant Migration. Vertical migration of the following contaminants of potential concern were modeled in Scenario A:

- TBP
- Cs-137
- Co-60
- Sr-90
- Pu-238, Pu-239, Pu-239/240
- total U.

Each simulation was run until either the contaminant being modeled reached the unconfined aquifer at a concentration of 1 pCi/L, or it became apparent that it never would. In general those contaminants which were both immobile and short-lived never reached the water table at unacceptable levels.

Of the contaminants modeled, Cs-137 and Sr-90 are among the most immobile and shortest-lived. Modeling results confirmed that both Cs-137 and Sr-90 were bound by soil particulate in the upper 23 m (74 ft) at the time discharge to the 216-B-45 crib ceased. Downward migration of Cs-137 and Sr-90 was simulated following cessation of discharge, but radioactive decay prevented either contaminant from reaching the water table at significant concentrations.

Presented in Figures 5-15, 5-16, and 5-17 are relative saturations, streamlines, and pore-water concentrations for Cs-137 at the cessation of discharge to crib 216-B-45, 150 yr after discharge began, and 300 yr after discharge began, respectively. Migration of Sr-90 is shown in Figures 5-18, 5-19, and 5-20 in the same manner, but instead the maximum time shown is 750 yr after discharge began. The vertical plane shown in all figures is that closest to the crib centerline, where the greatest concentrations would be expected. Note that relative saturation is defined as:

$$\frac{\text{predicted moisture content - residual moisture content}}{\text{saturated moisture content - residual moisture content}}$$

where all values are expressed on a volumetric basis.

Although plutonium has a large distribution coefficient similar to that of Cs-137 and Sr-90, it also has a very long half life. Consequently an extremely long time is required before plutonium reaches the water table. Migration of plutonium is shown in Figures 5-21, 5-22, and 5-23, the latter of which is for 9,500 yr after discharge began. Diffusion is the major reason that plutonium reaches the water table as early as 9,500 yr into the future. The computer resources available were not adequate to simulate into the future far enough to predict the maximum concentration of plutonium reaching the water table. The breakthrough curves shown in Figure 5-24 does, however, suggest that plutonium concentrations in the unconfined aquifer could eventually be over 100 pCi/L (the current MCL is 15 pCi/L) in tens-of-thousands of years.

The contaminants TBP, Co-60, and Total U are highly mobile compared to Cs-137, Sr-90, and plutonium, and as such reach the water table at concentrations exceeding the 1 pCi/L (1 μg/kg for TBP) level. Migration of TBP is shown in Figures 5-25, 5-26, and 5-27 at the time of cessation of discharge, 37 yr after discharge began (corresponding to 1992), and 150 yr after discharge began, respectively. Migration of Co-60 is shown at discharge cessation, 37 yr after discharge began (1992), and 100 yr after discharge began in Figures 5-28, 5-29, and 5-30. Total U migration at the cessation of discharge, 3 yr after discharge began, and 150 yr after discharge began is shown in Figures 5-31, 5-32, and 5-33.

Scenario B Contaminant Migration. Vertical migration of the following contaminants of potential concern were modeled in Scenario B:

- TBP
- Co-60
- Nitrate
- Tc-99
- total U

Because plutonium migration in Scenario A was a diffusion dominated process, results from modeling plutonium in this scenario would be similar to those of Scenario A. If instead plutonium migration was an advection or mechanical dispersion dominated

process, then modeling results would be highly dependent upon the difference in recharge due to meteoric infiltration between the two scenarios.

Migration of TBP at the cessation of disposal, 37 yr after disposal began (1992), and 300 yr after disposal began is shown in Figures 5-34, 5-35, and 5-36, respectively. Only relative concentrations (to unity) were modeled for TBP since there was no inventory of TBP disposal in 200-BP-1 cribs. Although adsorption prevented TBP from reaching the water table after 300 yr, eventually the model would predict TBP entering the unconfined aquifer. The concentration (95% UCL) of TBP in the crib infiltration gravels/soils is 16,000 μg/Kg; therefore, assuming a K_d for TBP of 4, the associated maximum pore water concentration would be about 4000 μg/L (assuming the Freundlich isotherm is linear). At 300 years (Figure 5-36), TBP pore water concentration at 15 meters (50 ft.) above the water table is seven orders-of-magnitude less than the maximum pore water concentration in the crib infiltration zone. These results indicate that if TBP will ever impact the water table above detectable concentrations, it would probably occur in excess of 1000 yr. TBP, being an organic compound, is expected to degrade somewhat in the time predicted to reach the water table and is not expected to cause major impacts to the aquifer.

Migration of Co-60 at cessation of disposal, 5 yr after disposal began, and 37 yr after disposal began (1992) is shown in Figures 5-37, 5-38, and 5-39, respectively. Although Co-60 has been detected in the unconfined aquifer it did not reach the water table in this simulation. As such the distribution coefficient was decreased from 1.94 to 0.5 mL/g (0.232 to 0.06 gal/lb) and the simulation rerun. Migration of Co-60 using the lower distribution coefficient is shown in Figures 5-40, 5-41, and 5-42 at cessation of disposal, 37 yr after disposal began (1992), and 100 yr after disposal began, respectively. After 100 yr, the Co-60 has spread and decayed so that the concentration within the saturated zone is less than 1 pCi/L. The breakthrough curves shown in Figure 5-43 also illustrate that the maximum Co-60 concentrations in the saturated zone beneath the 200-BP-1 operable unit were reached shortly after crib operation began and will continue to decline in the future. The modeling results are consistent with the observed groundwater monitoring in that a more concentrated slug of Co-60 entered the aquifer in the past than is currently occurring from sources in the 200-BP-1 operable unit (Appendix E).

A unit pCi/L source concentration was assumed for Tc-99 due to the lack of inventory information. Thus explaining the maximum concentration of 1 pCi/L in Figures 5-44, 5-45, and 5-46, which depict migration of Tc-99 at cessation of disposal, 37 yr after disposal began (1992), and 300 yr after disposal began, respectively. Although the relative vadose zone distribution of Tc-99 shown in the aforementioned figures may be accurate, the actual Tc-99 pore water concentrations are likely much greater as demonstrated by groundwater concentrations exceeding 600 pCi/L directly downgradient of crib 216-B-45. The breakthrough curve of Figure 5-47 indicate Tc-99 reached groundwater at maximum concentrations in the late 1970s, with a gradual decline in groundwater concentrations over most of the remaining simulated 300 yr. It is probably more likely that maximum groundwater concentrations were reached shortly after cessation of disposal, and that the decline in groundwater concentration with time was and will be more rapid. This is evidenced by the maximum Tc-99 groundwater concentrations (Appendix E) being an order-of-magnitude greater downgradient (≈15000 pCi/L) than beneath the 200-BP-1 operable unit (≈400-700 pCi/L). Technetium-99 migration, that is more representative of actual behavior, could probably be simulated using hydraulic properties of the vadose zone that permits faster drainage of transient moisture slugs such as from crib effluent disposal.

Total U migration at cessation of disposal, 37 yr after disposal began (1992), and 150 yr after disposal began is shown in Figures 5-48, 5-49, and 5-50, respectively. The breakthrough curves shown in Figure 5-51 indicate that the maximum total U concentration in the saturated zone could exceed 3,000 pCi/L at approximately 4,500 yr in the future. Total U is shown reaching the water table at a concentration exceeding 1 pCi/L by about 37 yr after the start of crib operations. This is consistent with observations of groundwater monitoring (Appendix E), where uranium concentrations appear to be 1 to 3 pCi/L greater in downgradient groundwater than the Hanford Site background value of 3.4 pCi/L.

Migration of nitrate at cessation of disposal, 5 yr after disposal began and 37 yr after disposal began (1992) is shown in Figures 5-52, 5-53 and 5-54, respectively. Nitrate reached the water table shortly after cessation of the effluent discharges which reflect its mobile nature (assumed K_d of 0). The breakthrough curve (Figure 5-55) indicates nitrate reached groundwater at maximum concentrations in the 1970s, with a gradual decline (similar to Tc-99 simulations) in groundwater concentrations over most of the remaining simulated 300 years. The decline in groundwater concentrations beneath the cribs was probably more rapid that predicted. Groundwater monitoring in the 200-BP-1 operable unit indicates that nitrate concentrations increase from about 50 to 70 mg/l, immediately upgradient of the operable unit to 70 to 80 mg/l in wells immediately downgradient of the cribs (Appendix E), thus the cribs are probably contributing an additional 10 to 20 mg/l nitrate concentration to the unconfined aquifer. The model simulation predicts current nitrate impacts of about 1000 to 10,000 mg/l, much higher than observed. As mentioned for Tc-99 simulations, model predictions could possibly be improved using hydraulic parameters for vadose zone soils that would allow more rapid drainage of a transient slug of water such as effluent discharges.

Model simulations are sensitive to transient drainage for the more mobile contaminants such at Tc-99, Co-60, and nitrate. Because of a short half life, the differences in model predicted impacts to monitoring observations of Co-60 were not as apparent as for the more persistent Tc-99 and nitrate. Increasing drainage in the vadose zone is expected to provide higher and earlier maximum model-predicted concentrations in the water table with a more rapid decrease in concentration after the maximum slug has entered the water table. Such a change could improve comparison in modeling results to monitored observations. Simulations of the less mobile contaminants should be relatively insensitive to transient drainage because the flux of infiltrating and recharging water has stabilized to steady-state conditions. After drainage of the effluent discharge occurs (whether it is 20 years or 70 years), the assumed infiltration/recharge rate controls the flux of water migrating through and entering the aquifer. Thus, contaminants that take hundreds to thousands of years to reach the water table, will be less effected by early transient drainage conditions that exists after cessation of crib discharges.

Scenario C Contaminant Migration. Vertical migration of uranium was the only contaminant simulated using Scenario C conditions. The other contaminants of concern that were modeled either were mobile and have already reached the water table or will not reach the water table even with extreme infiltration/recharge ratios in the foreseeable future (>1000 years). Uranium because of its moderate mobility and long half-life is more sensitive to the infiltration/recharge rate for predicting future impacts.

Migration of uranium in Scenario C at the cessation of disposal, five years after disposal and 37 years after disposal began (1992) is shown on Figures 5-56, 5-57, and 5-58,

respectively. These results, and the breakthrough curves to the water table shown in Figure 5-59, indicate that uranium reaches the water table more rapidly and at higher maximum concentrations than in Scenario B, which is expected when the assumed infiltration rate is increased. The maximum impact to groundwater in Scenario C is predicted to occur in about 2500 years as compared to about 5000 years in Scenario B. Since the infiltration/recharge rate was increased 10 fold in Scenario C from Scenario B, the time for maximum impacts to result only decreased by 1 fold.

Some uncertainty exists with regard to future uranium impacts to the unconfined aquifer from the 200-BP-1 sources. Currently, groundwater monitoring indicates little, if any, uranium has impacted the unconfined aquifer (Appendix E). Observed groundwater uranium concentrations in the study area are barely above Hanford Site background concentrations. The results of Scenario A uranium simulations clearly illustrate the overly conservative input parameters which predict much higher uranium impacts by 1992. Scenario B is difficult to compare with monitored observation since neither have slightly elevated concentrations of uranium in the aquifer over background. Based on the distribution of uranium with depth in the crib and underlying vadose zone soils (Appendix A), uranium appears to be migrating less than simulation B predicts. It is feasible that uranium has a slightly higher K_d than was used in the simulations (K_d=1) or that the infiltration/recharge rate could also be less than the rate (1 cm/yr) assumed for Scenario B. Uranium has had reported K_d's as high as 3 for the Hanford site. The time necessary for the peak uranium impacts to the unconfined aquifer should be approximately proportional to the K_d. Therefore, changing uranium K_d from one (1) to three (3) should increase the time for maximum impact by approximately a factor of three (3). Overall, the uranium modeling results of Scenario B appear to adequately represent current conditions and predict future fate and transport.

Scenario D Moisture Migration. Results of Scenario D are presented in Figure 5-60 as relative saturation in a vertical plane bisecting crib 216-B-50 in the north-south direction. Since crib 216-B-50 received the largest volume of liquid effluent in the crib grouping 216-B-43 through 216-B-50, the maximum amount of lateral moisture spreading would be expected to occur around it. It is apparent from observation of Figure 5-60 that the northern boundary is too close to the 216-B-50 crib. An estimate of lateral moisture spreading to the north was instead determined by examining saturations south of the 216-B-50 crib, but is probably overconservative due to the moisture provided by cribs 216-B-46 and 216-B-49. Results indicate a maximum lateral moisture spreading of about 70 m (230 ft) to a depth of about 13 m (43 ft) sometime between 1974 and 1976. These results when extrapolated to contaminant spreading ignore any retardation or radioactive decay effects, and over predict the amount of spreading shown by field observations. Model refinement beyond that presented was deemed unnecessary at this time because any design of source control measures will require empirical observation to support modeling results. Information obtained from any future field investigations can alternatively be used for model refinement and calibration at a later date.

5.2.4 Biotic Pathways

The potential exists for contaminant transport to and through the terrestrial media at and near the 200-BP-1 operable unit. The evaluation of such transport was taken into account in the baseline risk assessment (Chapter 6) because of potential harmful effects to

the wildlife community. Three bird species which may frequent the area, the Swainson's hawk, the loggerhead shrike, and the burrowing owl, are considered sensitive terrestrial indicators and were evaluated with respect to their potential for being harmed by the operable unit contamination.

Contaminant transport through the 200-BP-1 operable unit ecosystem was difficult to characterize due to the lack of system and species-specific data. Conservative simplifying assumptions and surrogate uptake factors were used in the terrestrial biological transport pathway evaluations which follow. While the results are useful in assessing potential risks in Chapter 6, they must be regarded as semiquantitative or with a high degree of uncertainty.

5.2.4.1 Terrestrial Biological Transport. Contaminants of potential concern within the surface soil column (0 to 4.6 m or 15 ft) of the 200-BP-1 operable unit are Cs-137, Ra-226, Sb, Sr-90, Th-228, and total U. Soil sample analyses from two areas within the operable unit (cribs 216-B-43 through 216-B-50 and 216-B-57) were available, and the greater of the two UCL values for each contaminant was used in the calculations. The soil concentration values used are as follows:

Contaminant	Soil Concentration	
Cs-137	23	pCi/g
Ra-226	2.0	pCi/g
Sr-90	1.8	pCi/g
Th-228	0.74	pCi/g
Total U	1.4	pCi/g

Two generalized contaminant transport pathways for the species of concern were considered and can be represented as:

- Soil to vegetation → Great Basin pocket mouse → Swainson's hawk, loggerhead shrike, and burrowing owl.
- Soil → insects → Swainson's hawk, loggerhead shrike, and burrowing owl.

In the first model above, it was assumed that the birds ate Great Basin pocket mice exclusively. In the second, that the birds' diet were entirely insects. In nature, Swainson's hawks eat a combination of insects and rodents, as well as lizards and toads. Loggerhead shrike eat a combination of insects, rodents, small birds, and reptiles. Burrowing owls eat a combination of insects, rodents, and occasionally small birds and reptiles (Terres 1980).

In calculating the dose to the Great Basin pocket mice, it was assumed that its diet was 100% vegetation, with a dry to wet vegetation conversion factor of 0.4. Plant/soil transfer coefficients were taken from the open literature summarized in Chapter 6. Where possible, values from Hanford Site studies were used. Otherwise, we used the highest transfer value identified was used with transfer values are summarized below:

Contaminant	Transfer Coefficient
Cs-137	0.62
Ra-226	0.1
Sr-90	24
Th-228	0.0001
Total U	1

No insect intake factors are known to exist for the operable unit contaminants of potential concern. One study indicated a grasshopper uptake factor of approximately one (on a dry weight basis relative to soil concentrations) for dioxin (Paustenbach 1989). Therefore, this surrogate value is used for all contaminants of potential concern for the 200-BP-1 operable unit.

Home ranges for each species was determined by evaluating the percentage of time that the animal could potentially spend feeding within the site. This was determined by estimating the fraction of the site area within the receptor home range area. For organisms whose home range is smaller than the operable unit, it was assumed that 100% of their diet consisted of contaminated foodstuffs. This assumption was made for insects, Great Basin pocket mice, and jack rabbits. For organisms spending a fraction of their time feeding within the operable unit, a usage factor was calculated based on the proportion of their home range that the operable unit could encompass. For example, for a Swainson's hawk, whose home range is about 60 times larger than the 200-BP-1 operable unit, a usage factor was calculated by dividing the area of the operable unit (0.101 km^2 or 25.0 ac) by the hawks home range (5.77 km^2 or 1,430 ac). This usage factor was incorporated into the dose equation. In some cases a home range was derived from species densities observed on the Hanford Site. This is a reasonable derivation if the species of interest is territorial, as are the Swainson's hawk, loggerhead shrike, and the burrowing owl.

The estimated contaminant concentrations attributable to the operable unit within each type of prey (on a wet weight basis) were estimated (using equations 6-5 through 6-12 in Section 6.3) to be:

Insect:

Contaminant	Tissue Concentration	
Cs-137	6.9	pCi/g
Ra-226	0.62	pCi/g
Sr-90	0.54	pCi/g
Th-228	0.22	pCi/g
Total U	0.42	pCi/g

We assumed a dry to wet insect conversion factor of 0.3.

Great Basin pocket mouse:

Contaminant	Tissue Concentration	
Cs-137	93	pCi/g
Ra-226	9.3	pCi/g
Sr-90	1400	pCi/g
Th-228	2.8×10^{-6}	pCi/g
Total U	5.8	pCi/g

The uptake fractions for man (Baker and Soldat 1992) were used for the radionuclides because of the lack of any values for the species of concern. For antimony an animal/plant concentration factor of 0.01 was used.

The above tissue concentration values are used in Chapter 6 to assess the potential harm to local populations of the three sensitive bird species.

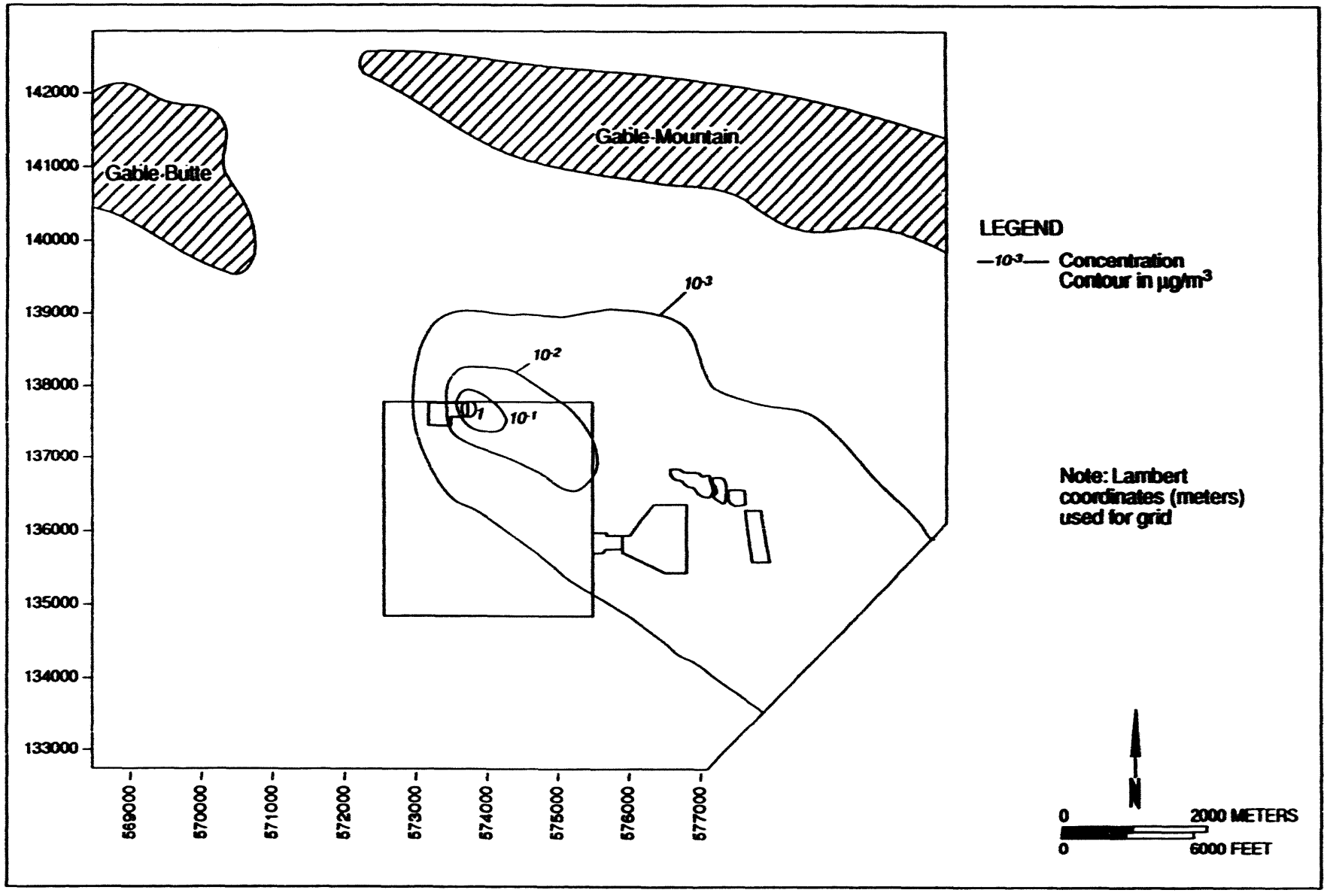


Figure 5-1. Concentration of Airborne Particulate from Near-Surface Soil Zone Emission Within the 216-B-43 Through 216-B-50 Crib Grouping Source Area.

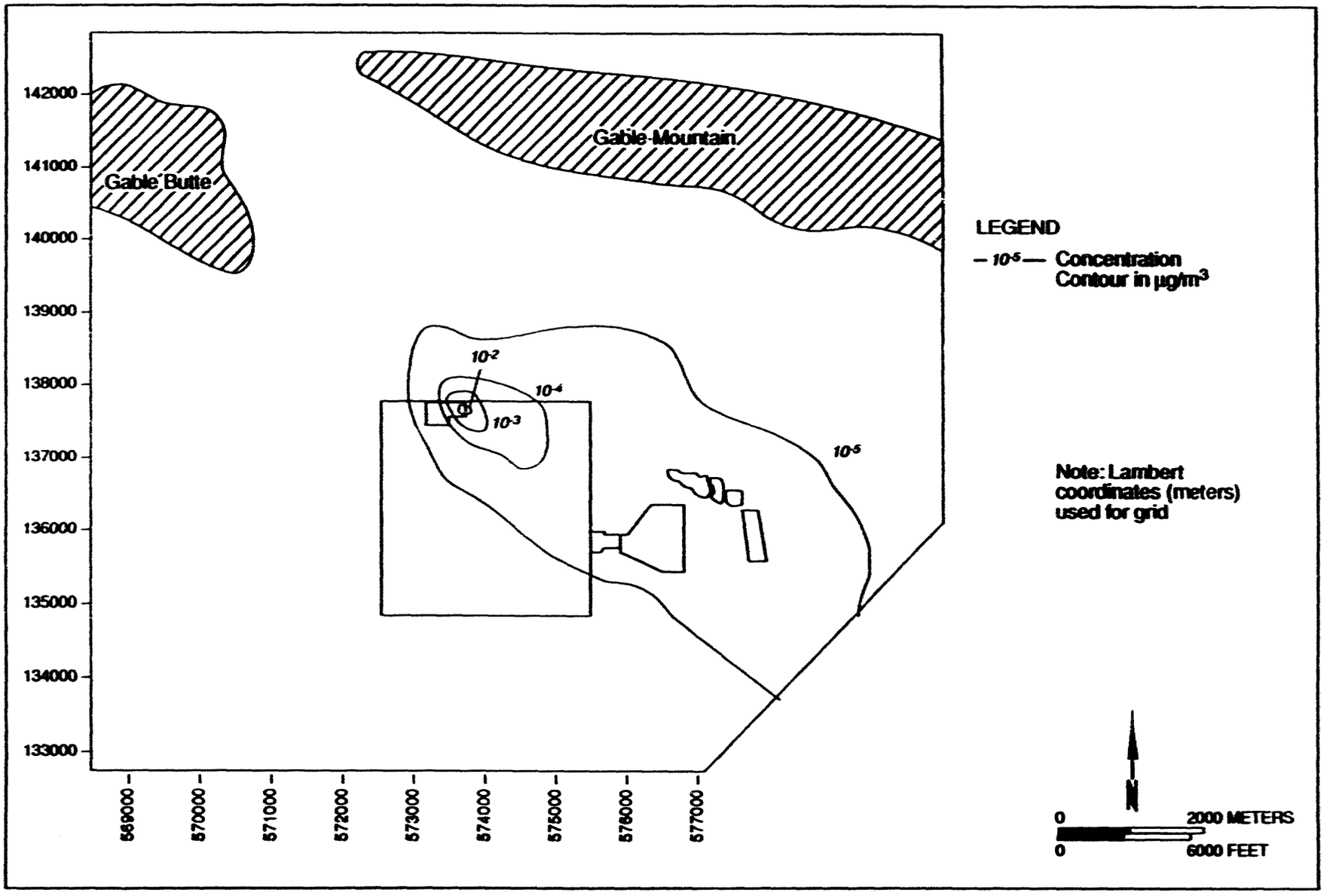


Figure 5-2. Concentration of Airborne Particulate from Subsurface Infiltration Gravel/Soil Zone Emission Within the 216-B-43 Through 216-B-50 Crib Grouping Source Area.

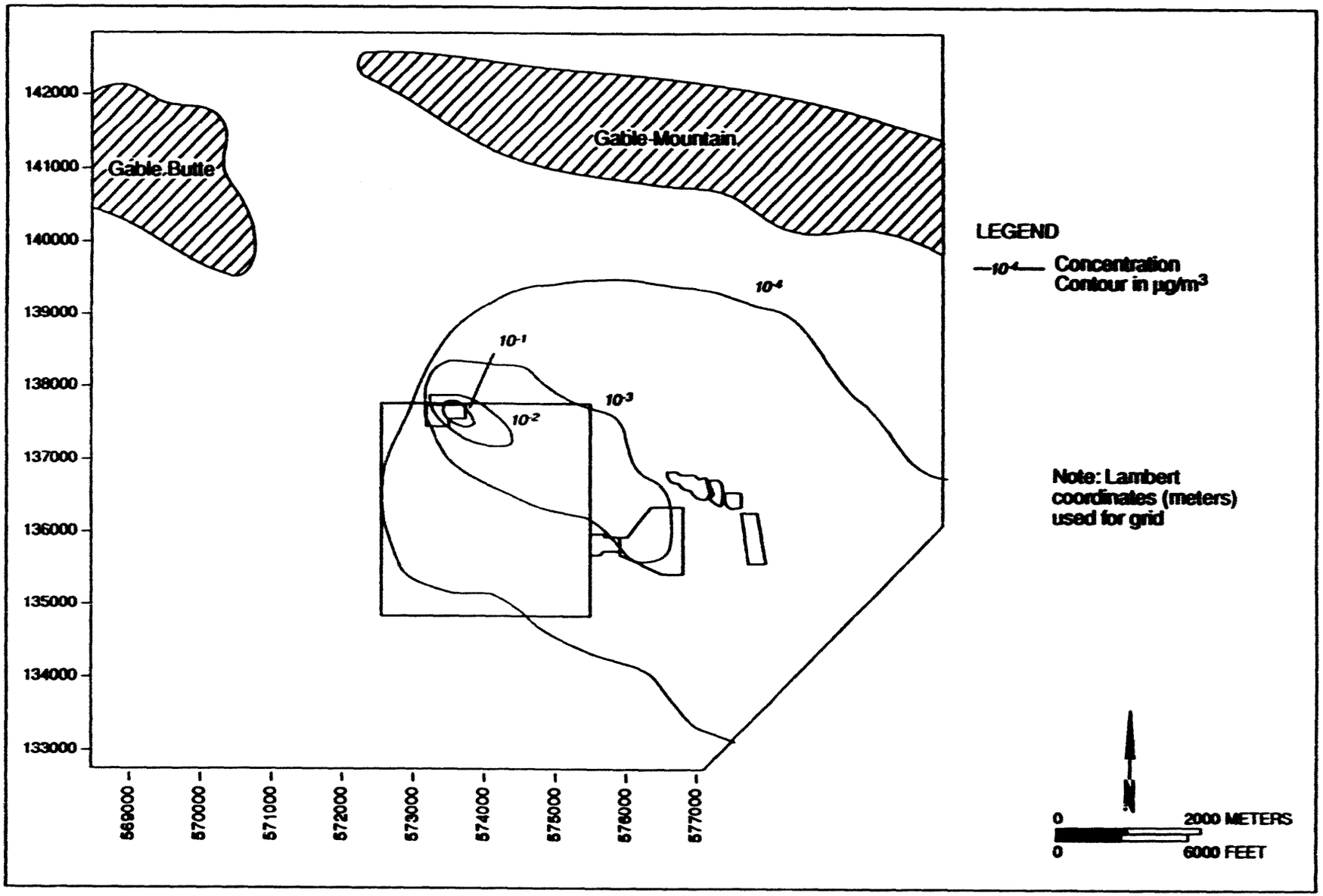


Figure 5-3. Concentration of Airborne Particulate from Near-Surface Soil Zone Emission Within the 216-B-57 Crib Source Area.

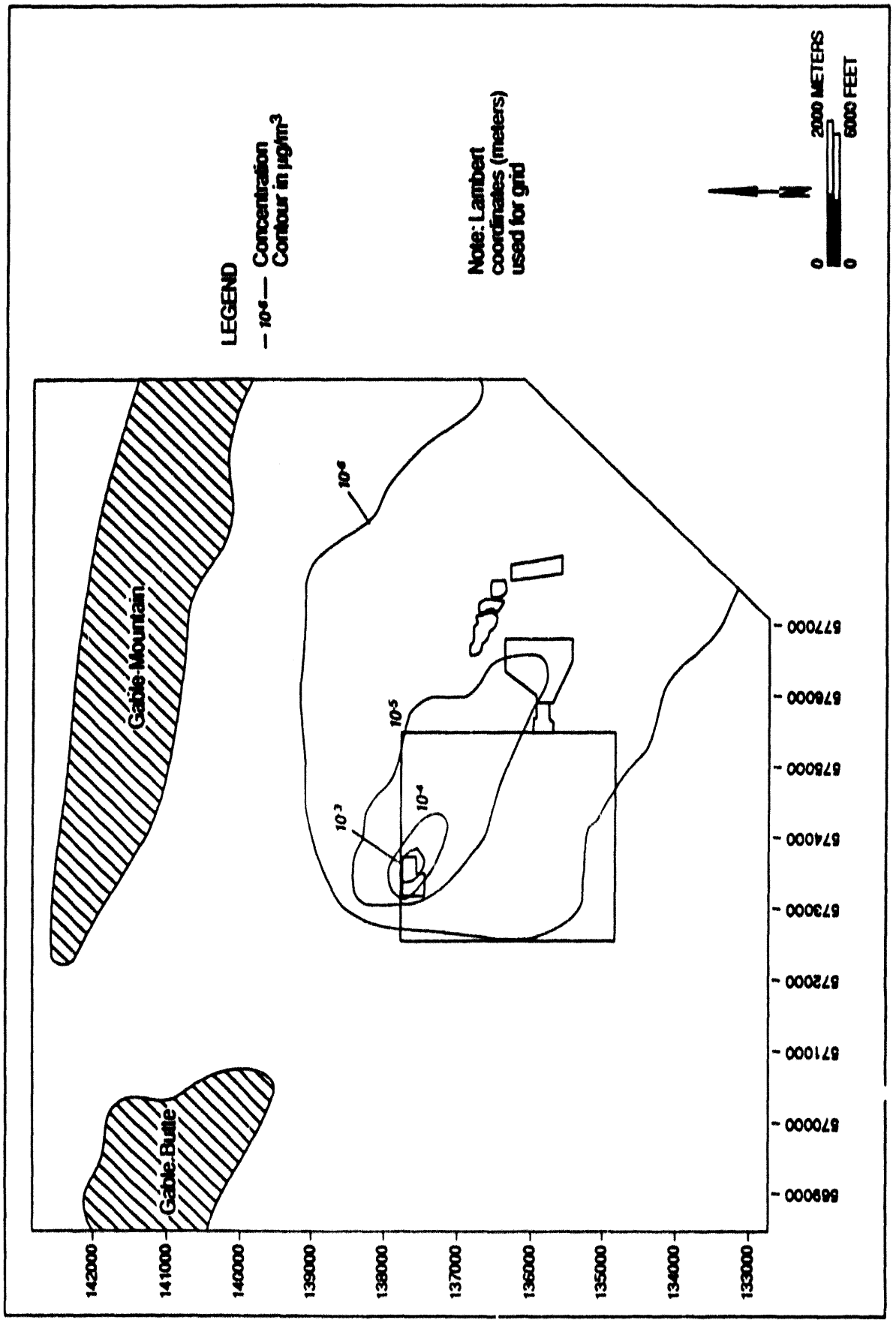


Figure 5-4. Concentration of Airborne Particulate from Subsurface Infiltration Gravel/Soil Zone Emission Within the 216-B-57 Crib Source Area.

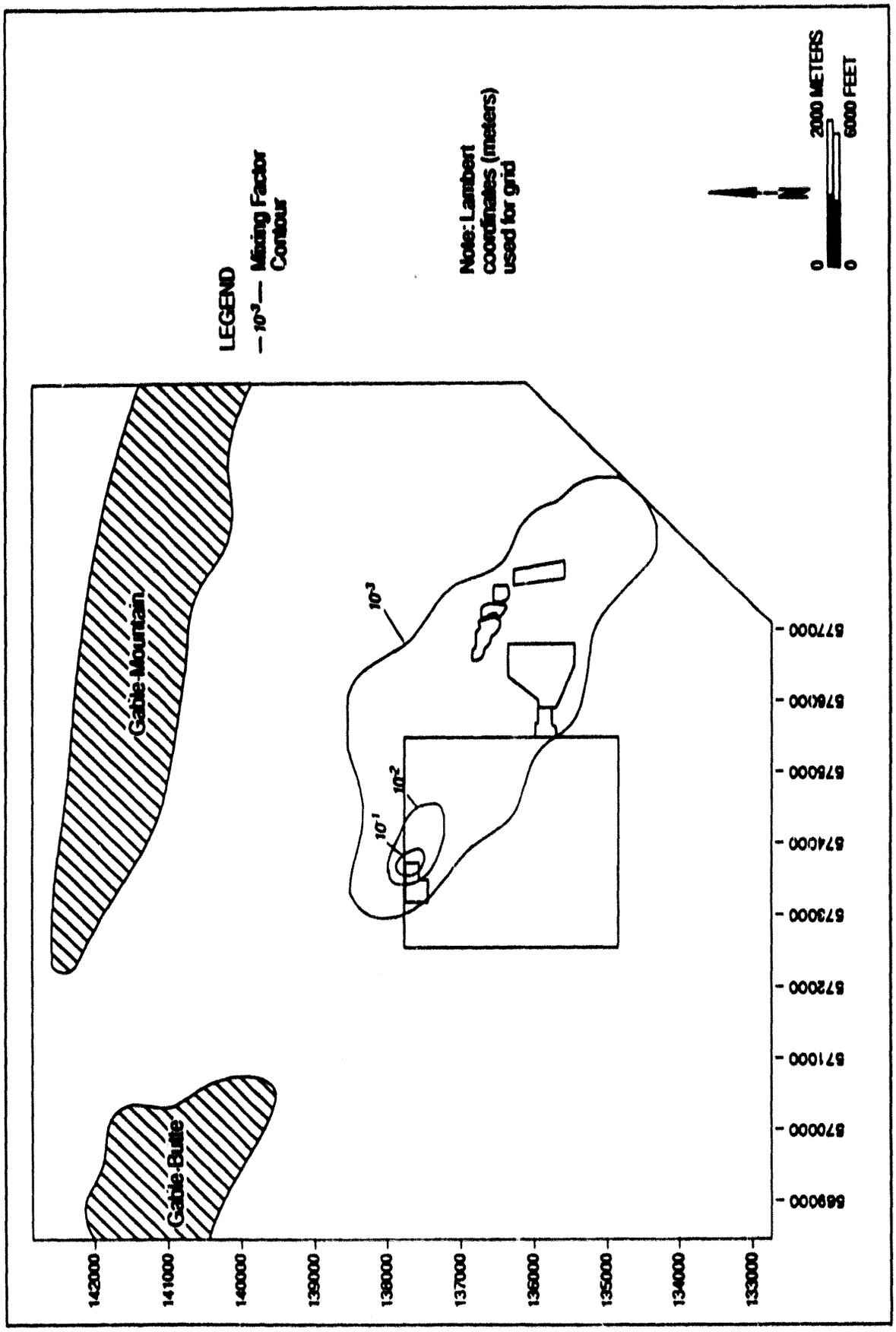


Figure 5-5. Mixing Factors for Ground Surface Concentrations from Near-Surface Soil Zone Emission Within the 216-B-43 Through 216-B-50 Crib Grouping Source Area.

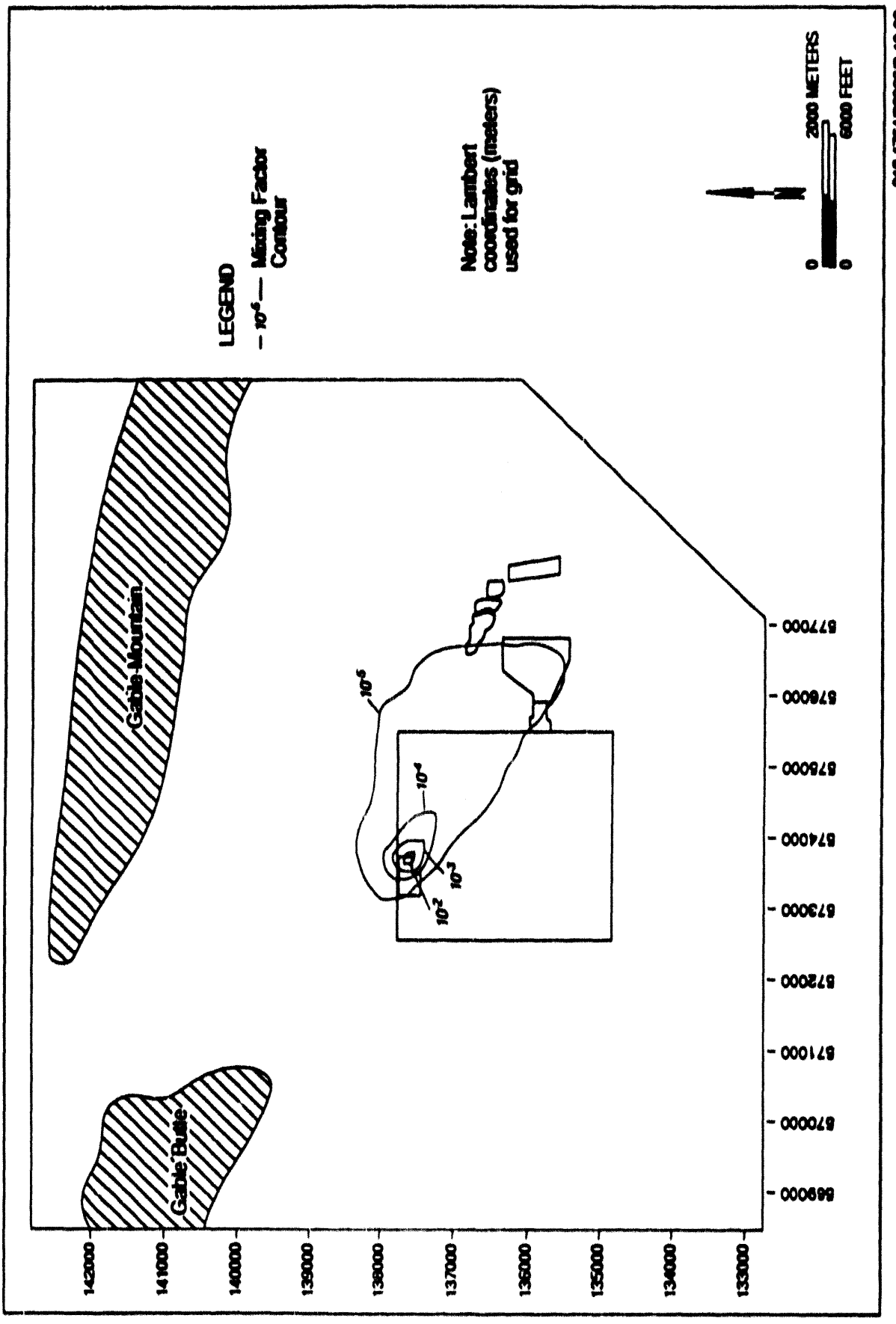


Figure 5-6. Mixing Factors for Ground Surface Concentrations from Subsurface Infiltration Gravel/Soil Zone Emission Within the 216-B-43 Through 216-B-50 Crib Grouping Source Area.

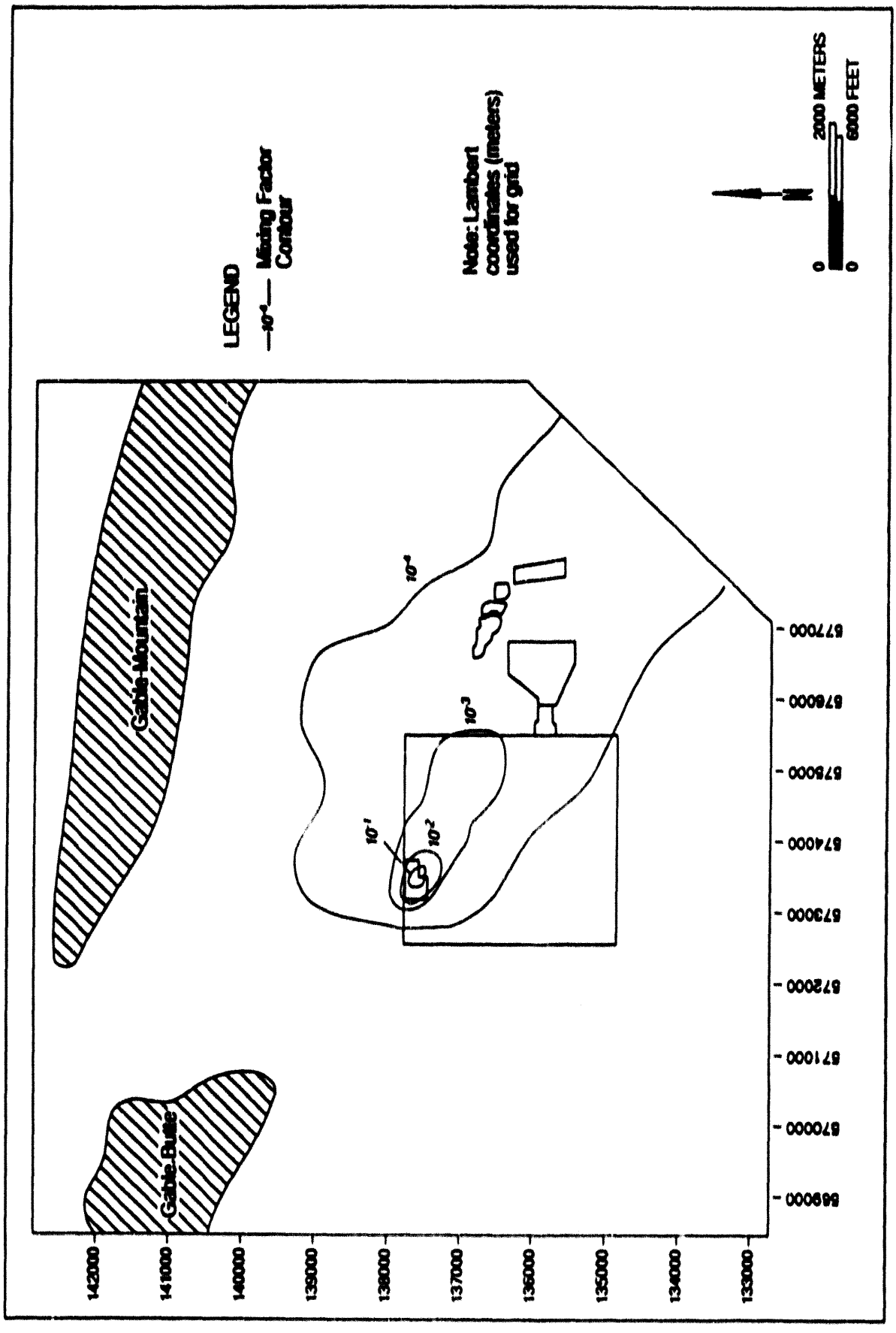


Figure 5-7. Mixing Factors for Ground Surface Concentrations from Near-Surface Soil Zone Emission Within the 216-B-57 Crib Source Area.

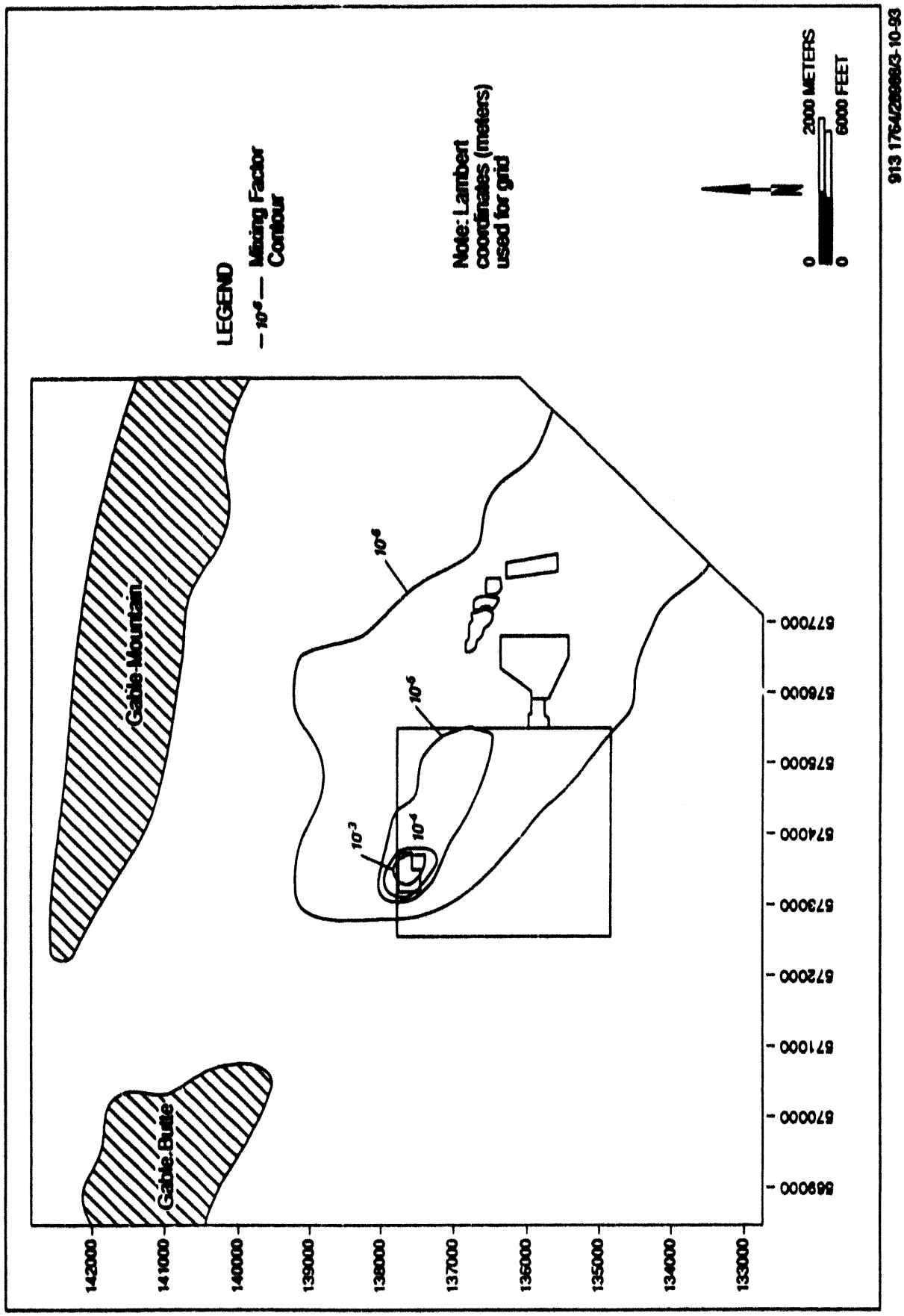


Figure 5-8. Mixing Factors for Ground Surface Concentrations from Subsurface Infiltration Gravel/Soil Zone Emission Within the 216-B-57 Crib Source Area.

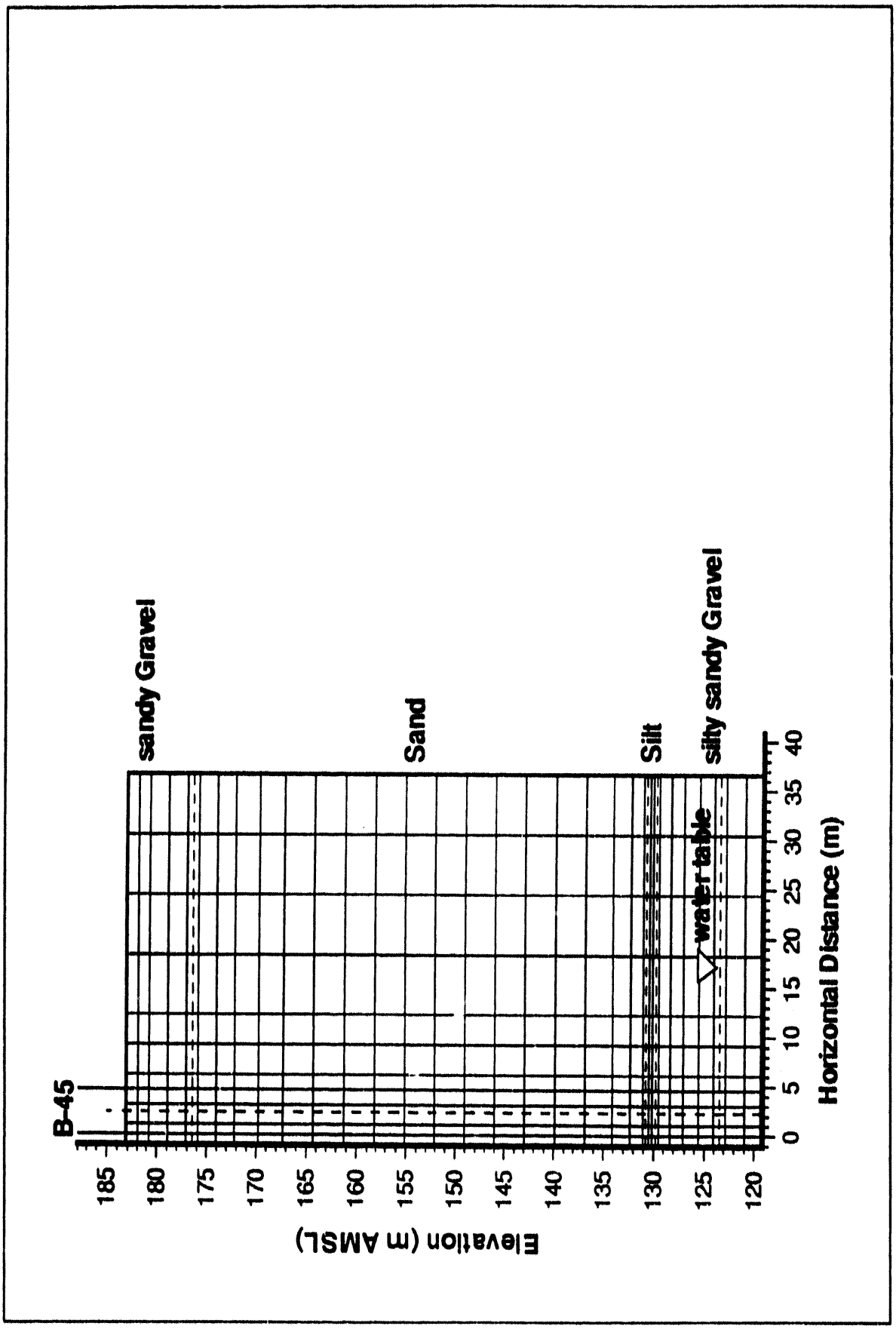


Figure 5-9. Finite Difference Mesh for Scenarios A, B and C.

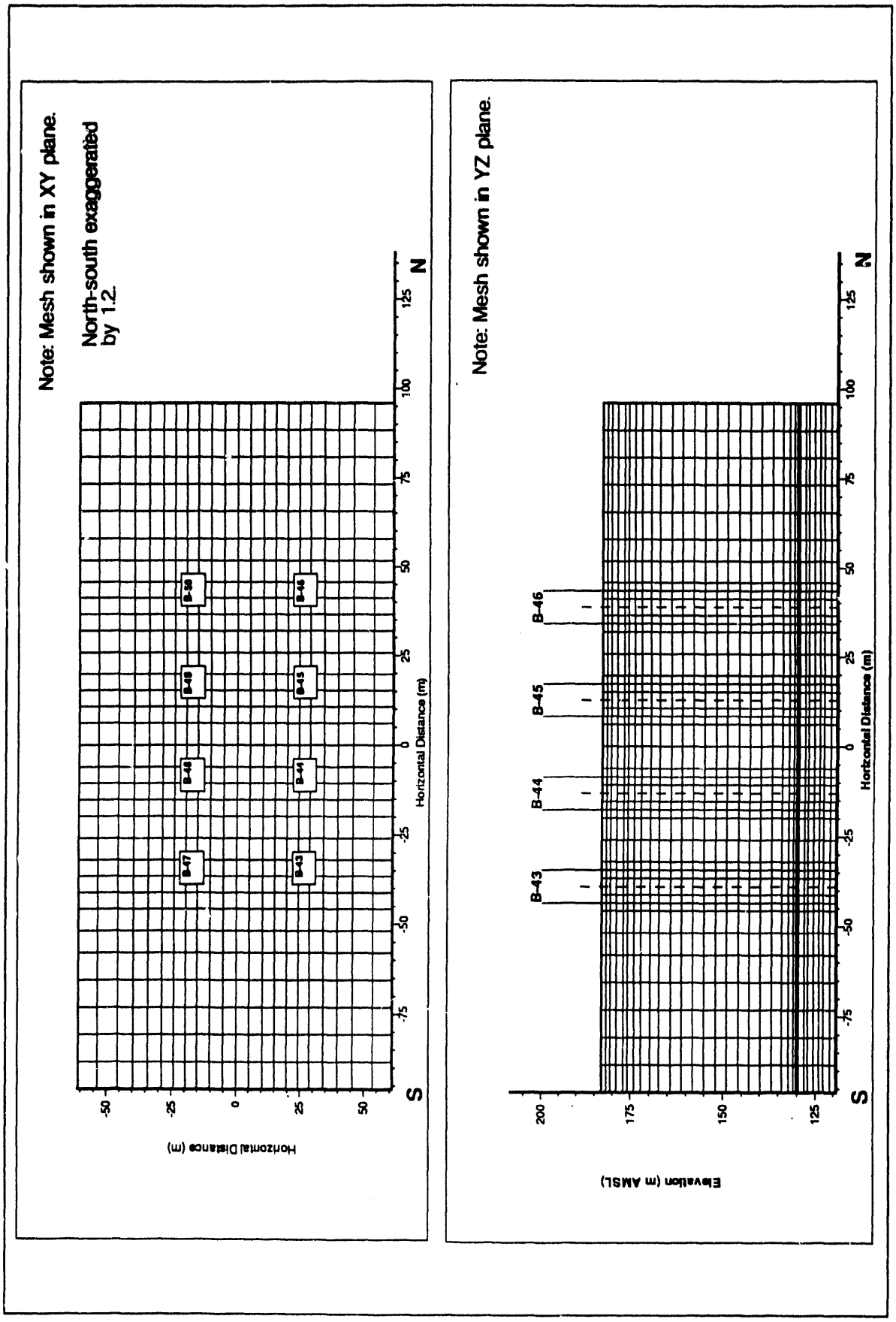


Figure 5-10. Finite Difference Mesh for Scenario D in Plan and Cross Section.

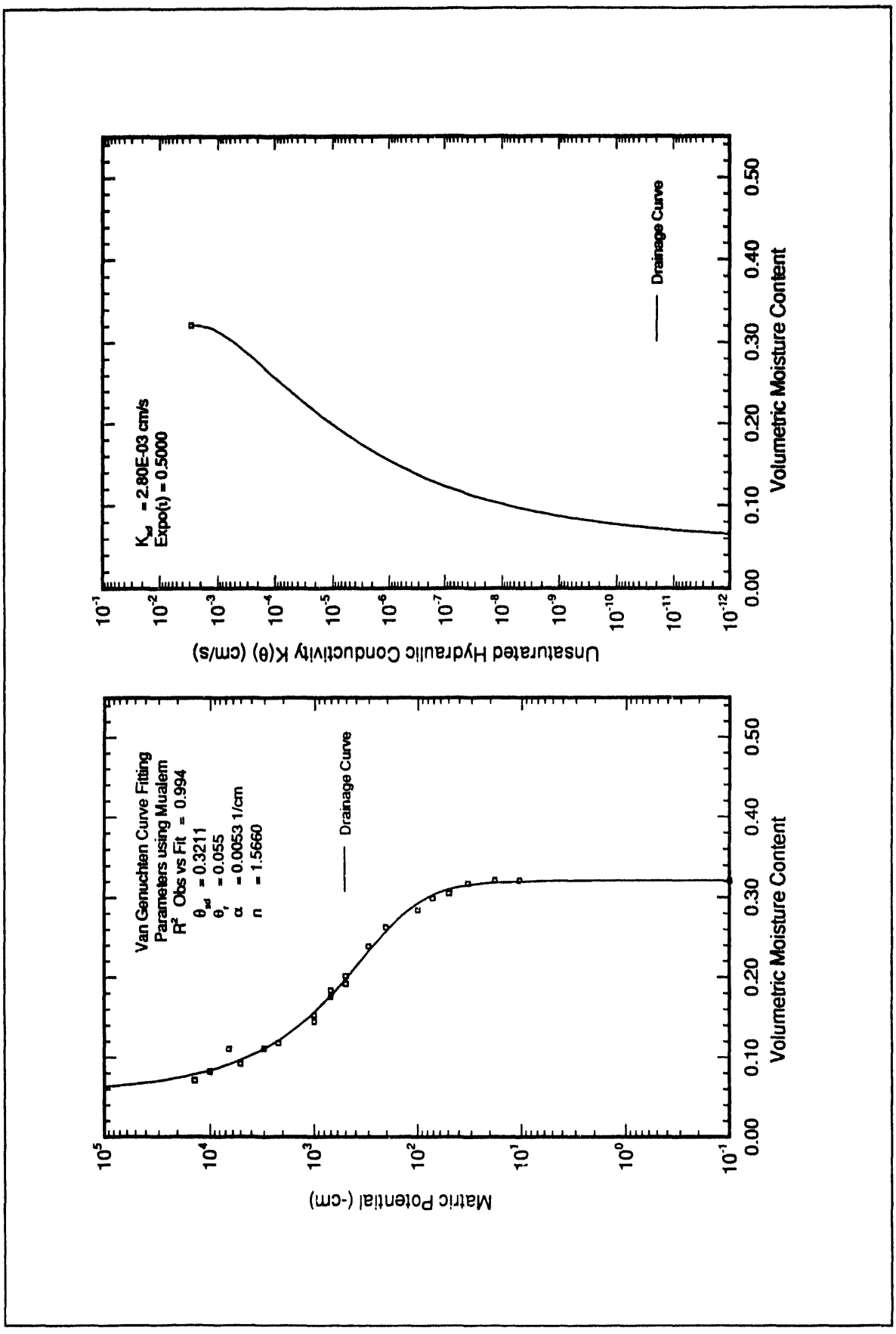
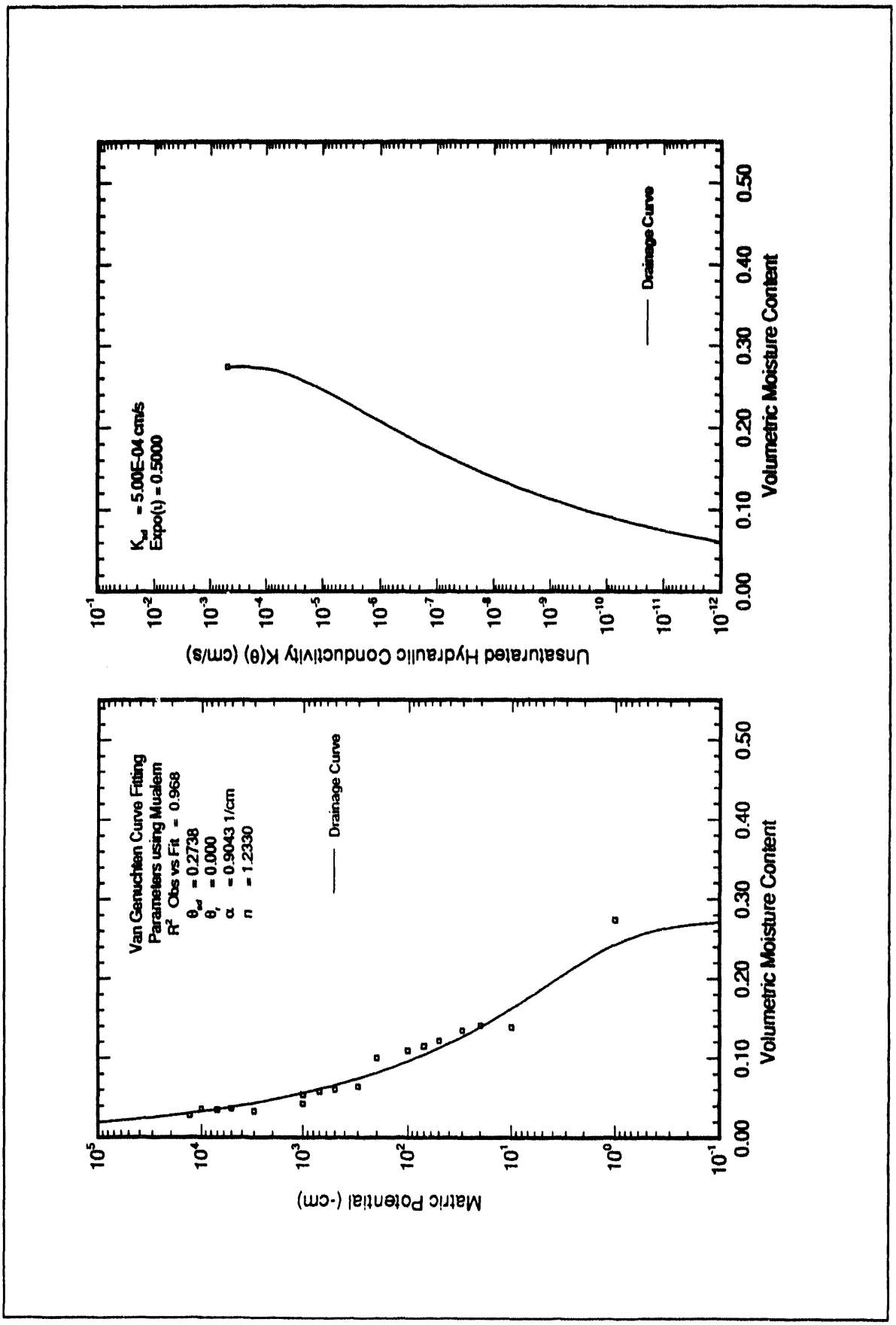


Figure 5-11. Sandy Gravel Matric Potential and Unsaturated Hydraulic Conductivity Curves.



913-1764/27012/12-30-92

Figure 5-12. Sand Matrix Potential and Unsaturated Hydraulic Conductivity Curves.

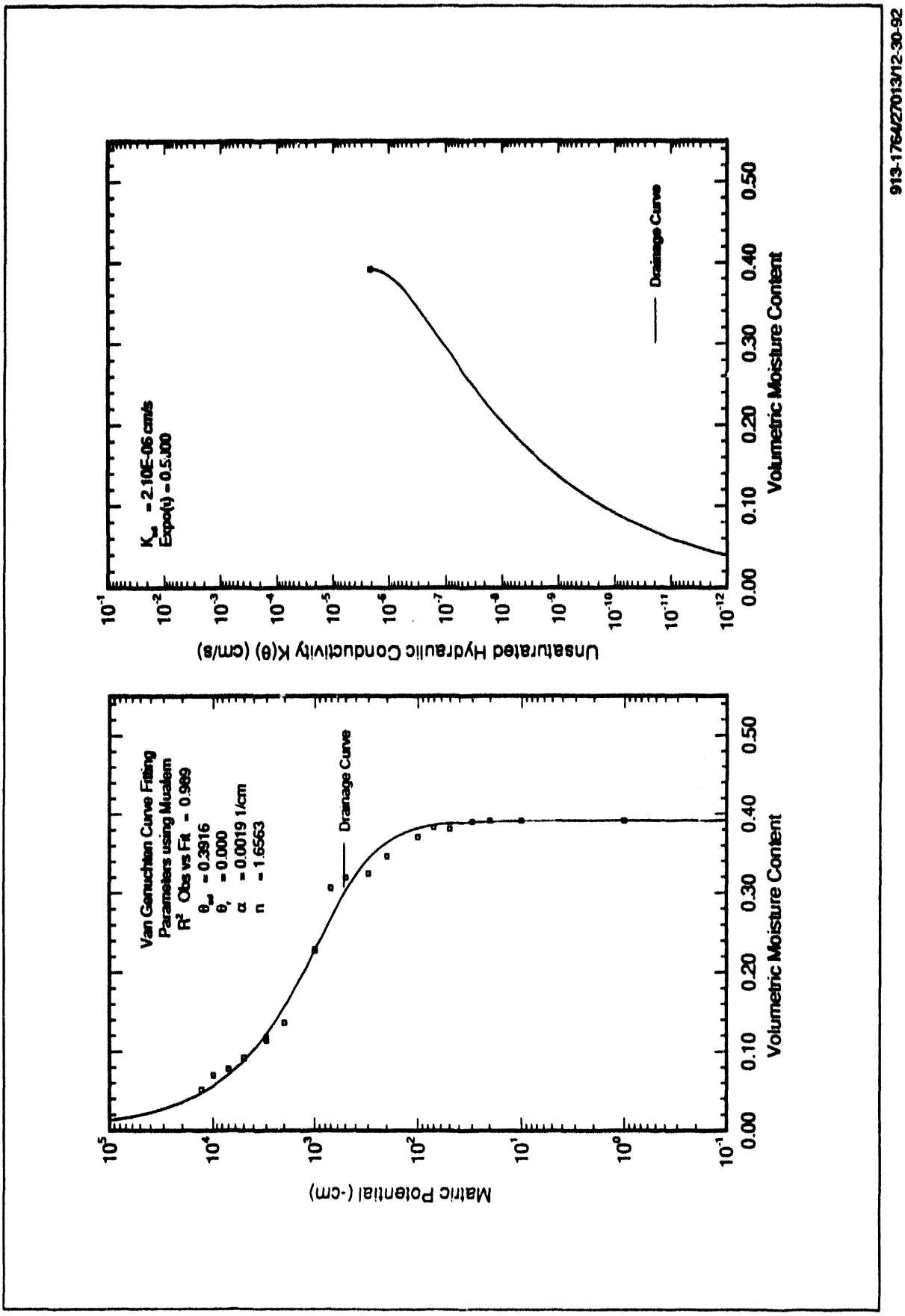


Figure 5-13. Silt Matrix Potential and Unsaturated Hydraulic Conductivity Curves.

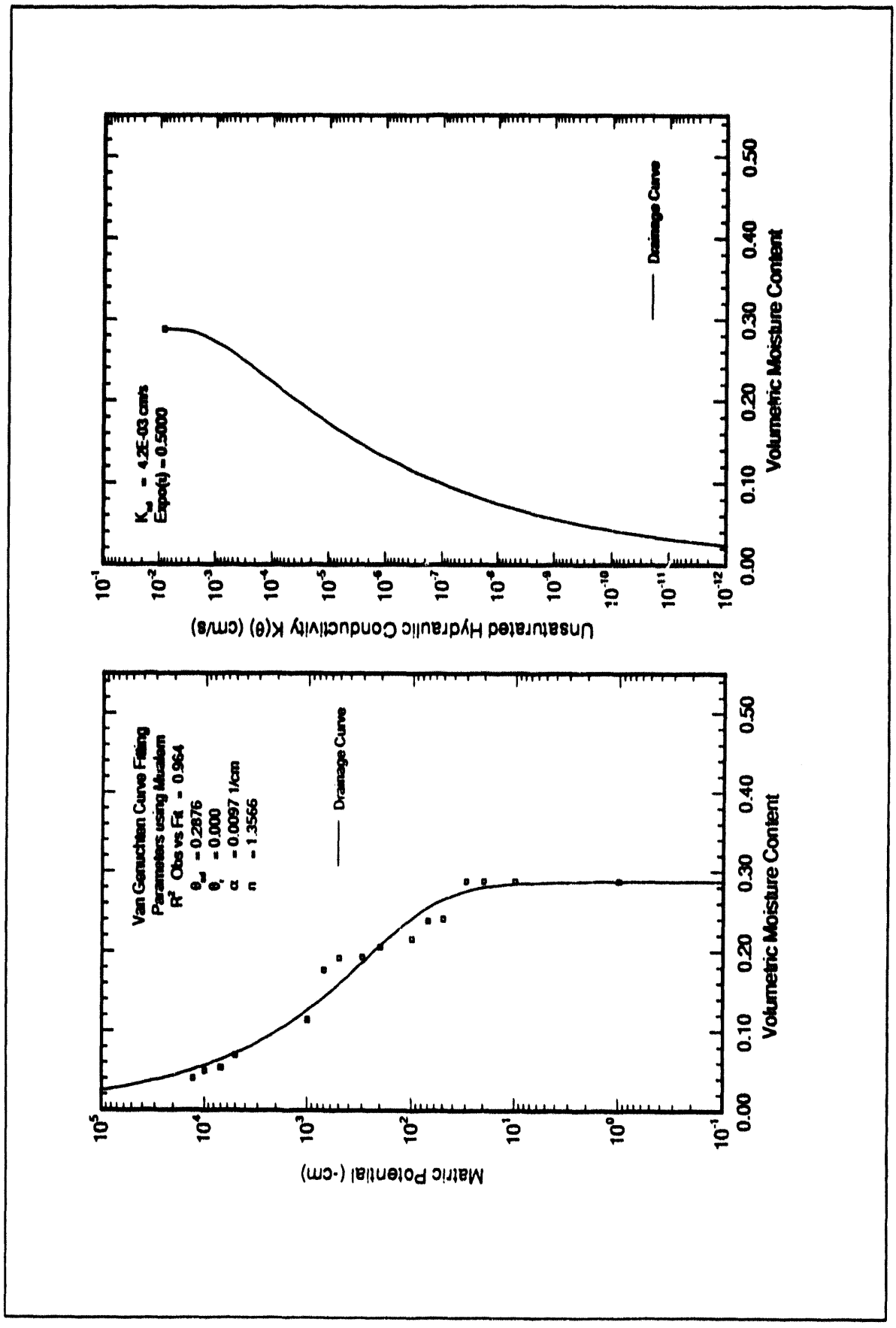
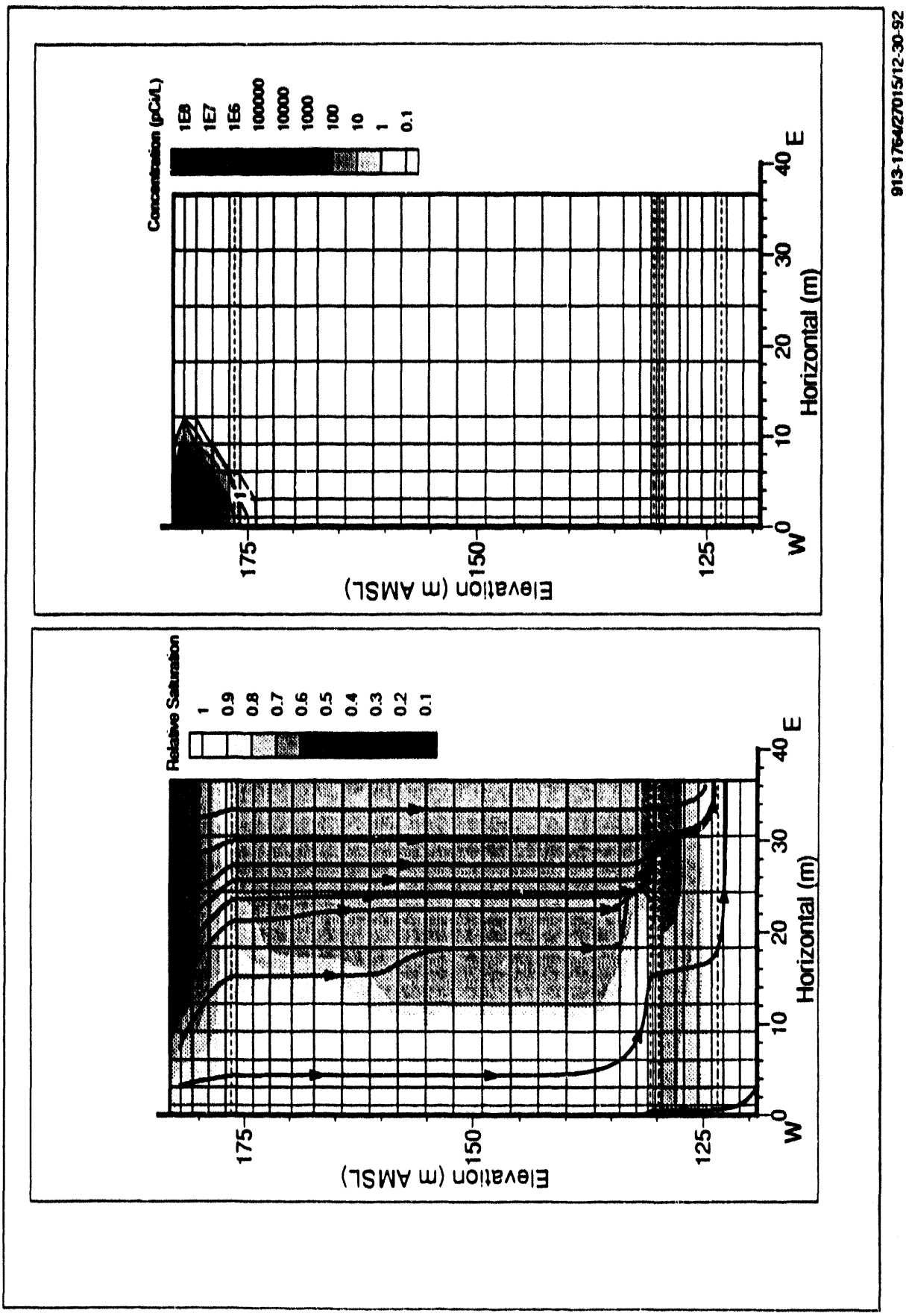
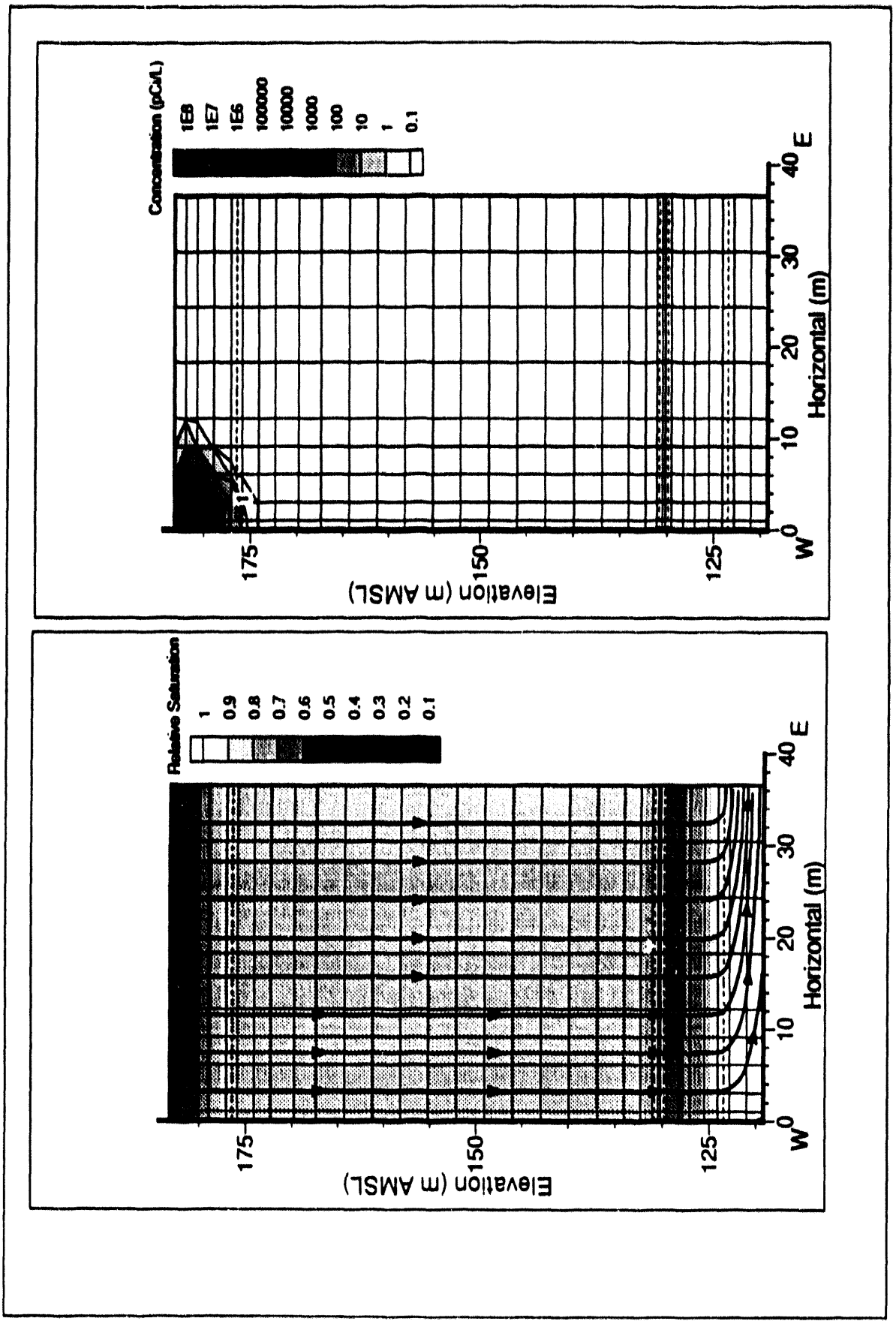


Figure 5-14. Silty Sandy Gravel Matric Potential and Unsaturated Hydraulic Conductivity Curves.



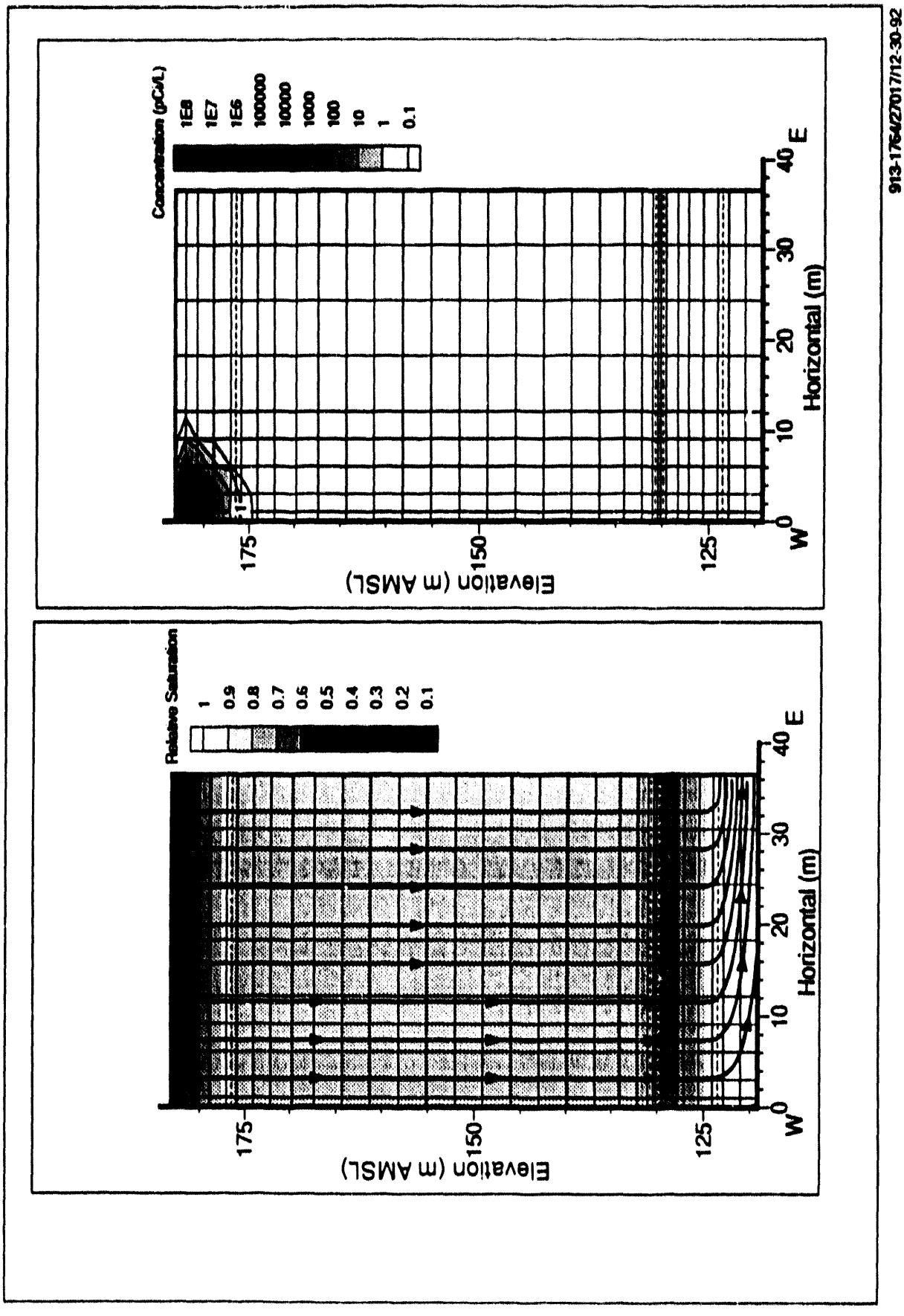
913-176/27015/12-30-92

Figure 5-15. Relative Saturation, Streamlines, and Cs-137 Concentration for Scenario A at Cessation of Discharge.



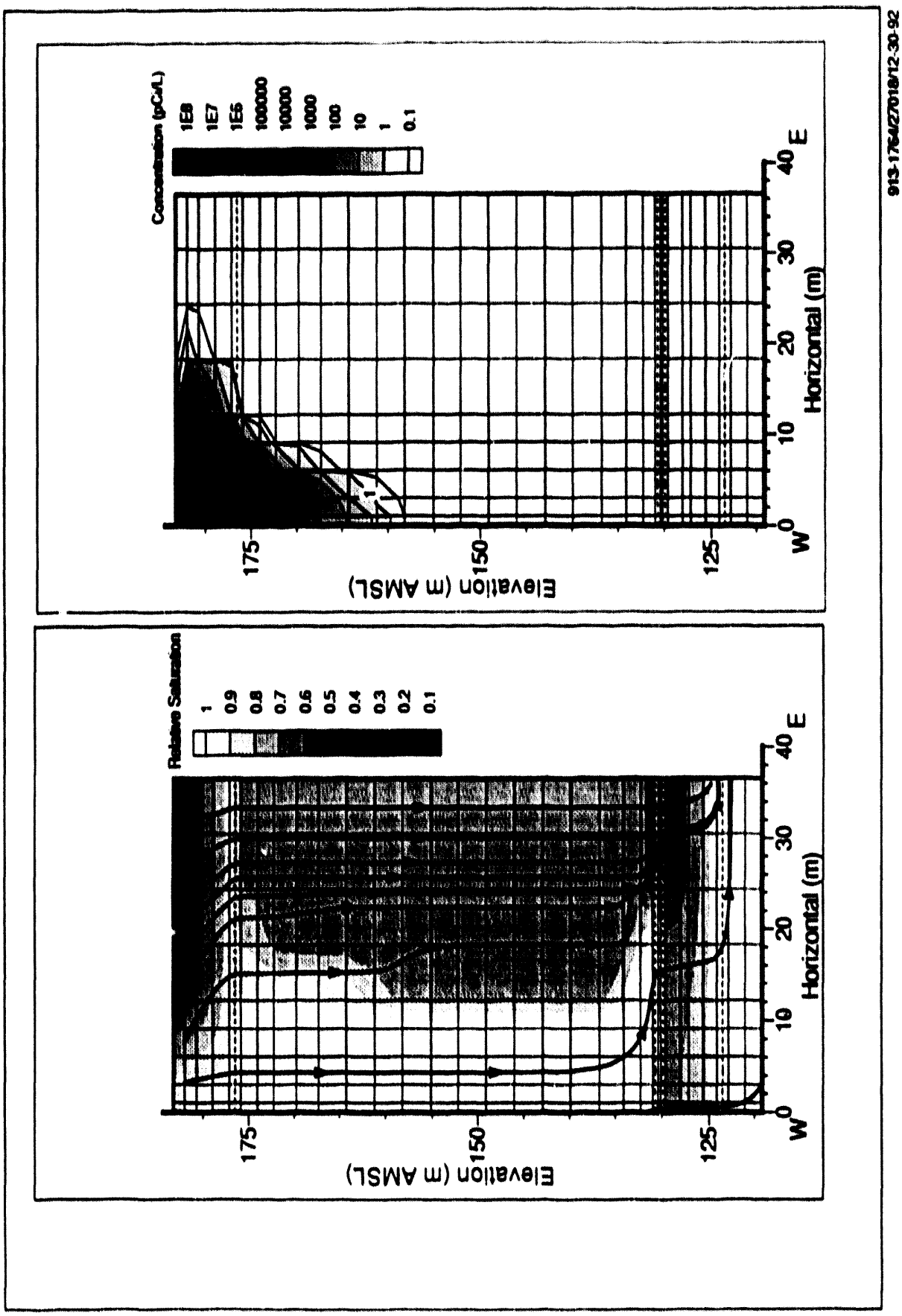
913-176/4/27016/12-30-92

Figure 5-16. Relative Saturation, Streamlines, and Cs-137 Concentration for Scenario A after 150 yr.



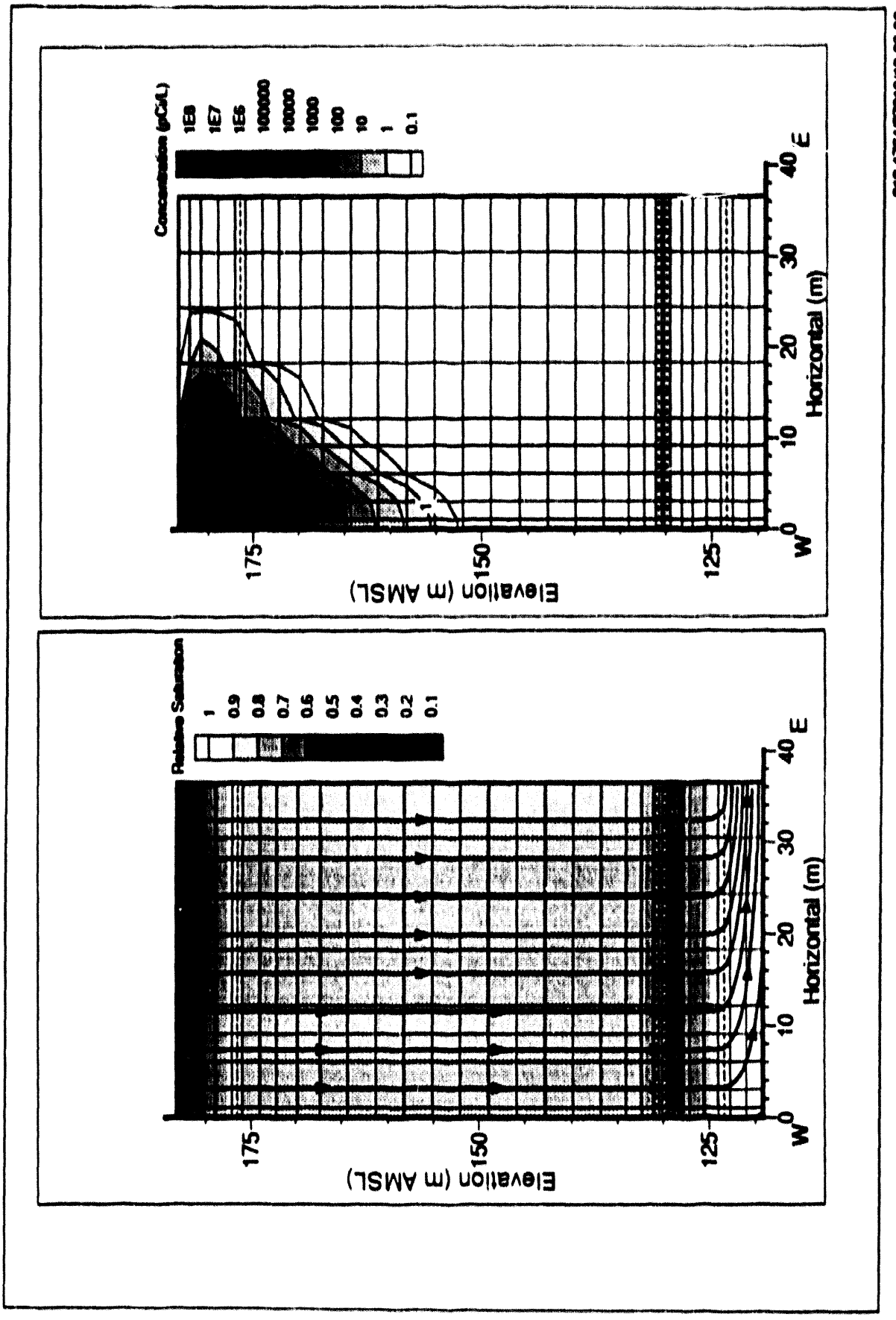
913-176/27017/12-30-92

Figure 5-17. Relative Saturation, Streamlines, and Cs-137 Concentration for Scenario A after 300 yr.



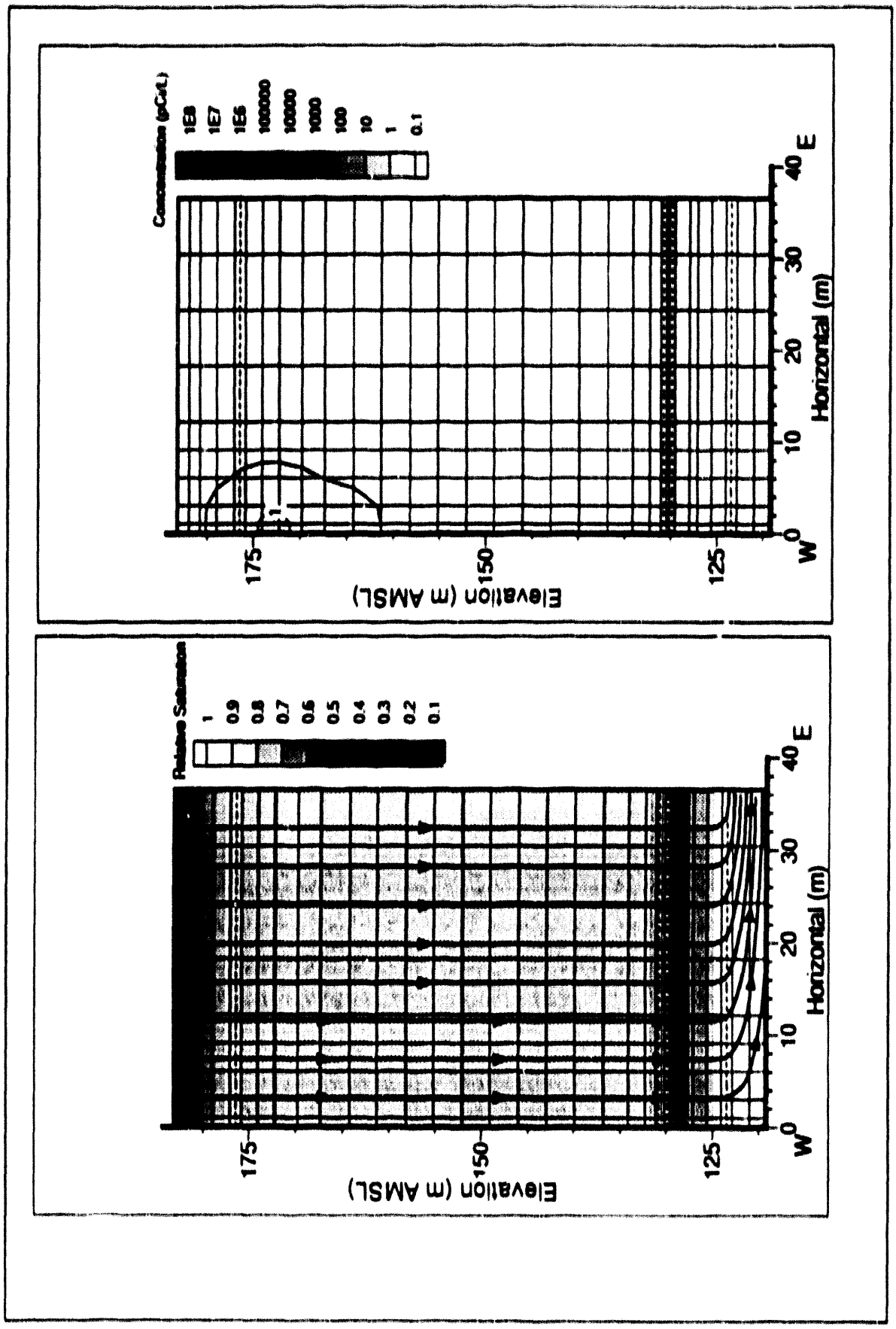
913-176/42701B/12-30-92

Figure 5-18. Relative Saturation, Streamlines, and Sr-90 Concentration for Scenario A at Cessation of Discharge.



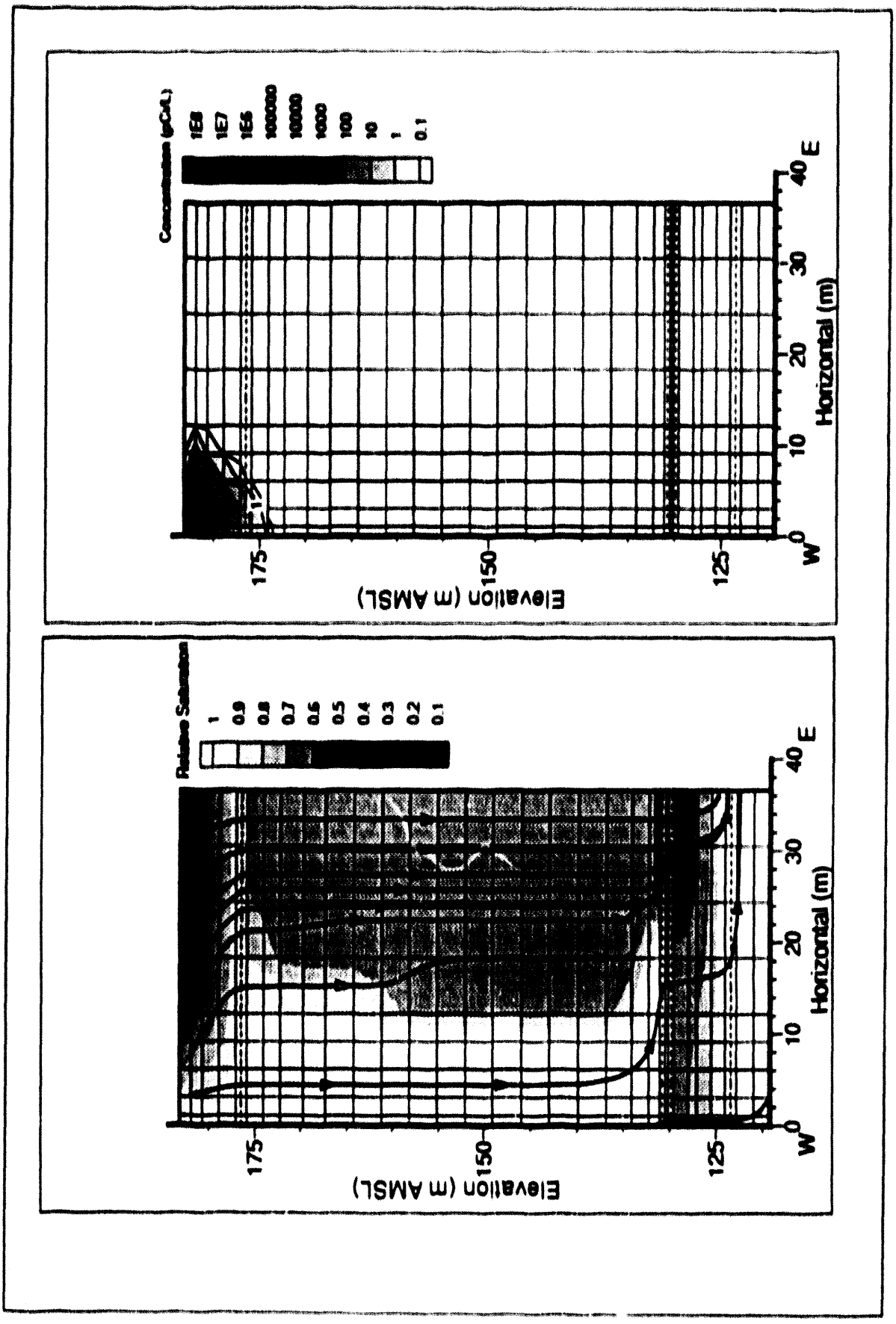
913-178/4Z7019/12-30-92

Figure 5-19. Relative Saturation, Streamlines, and Sr-90 Concentration for Scenario A after 150 yr.



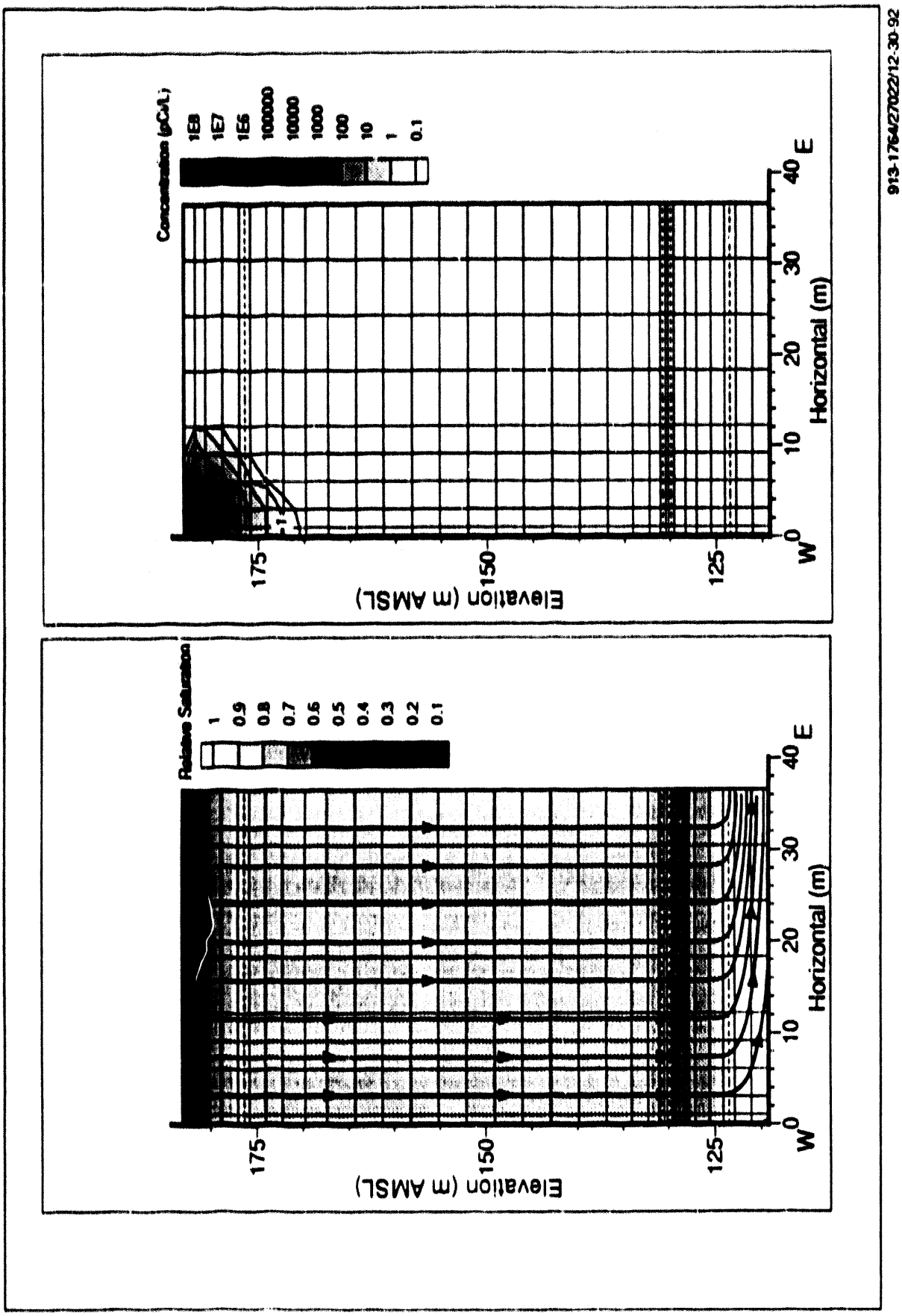
913-178/42702/012-30-92

Figure 5-20. Relative Saturation, Streamlines, and Sr-90 Concentration for Scenario A after 750 yr.



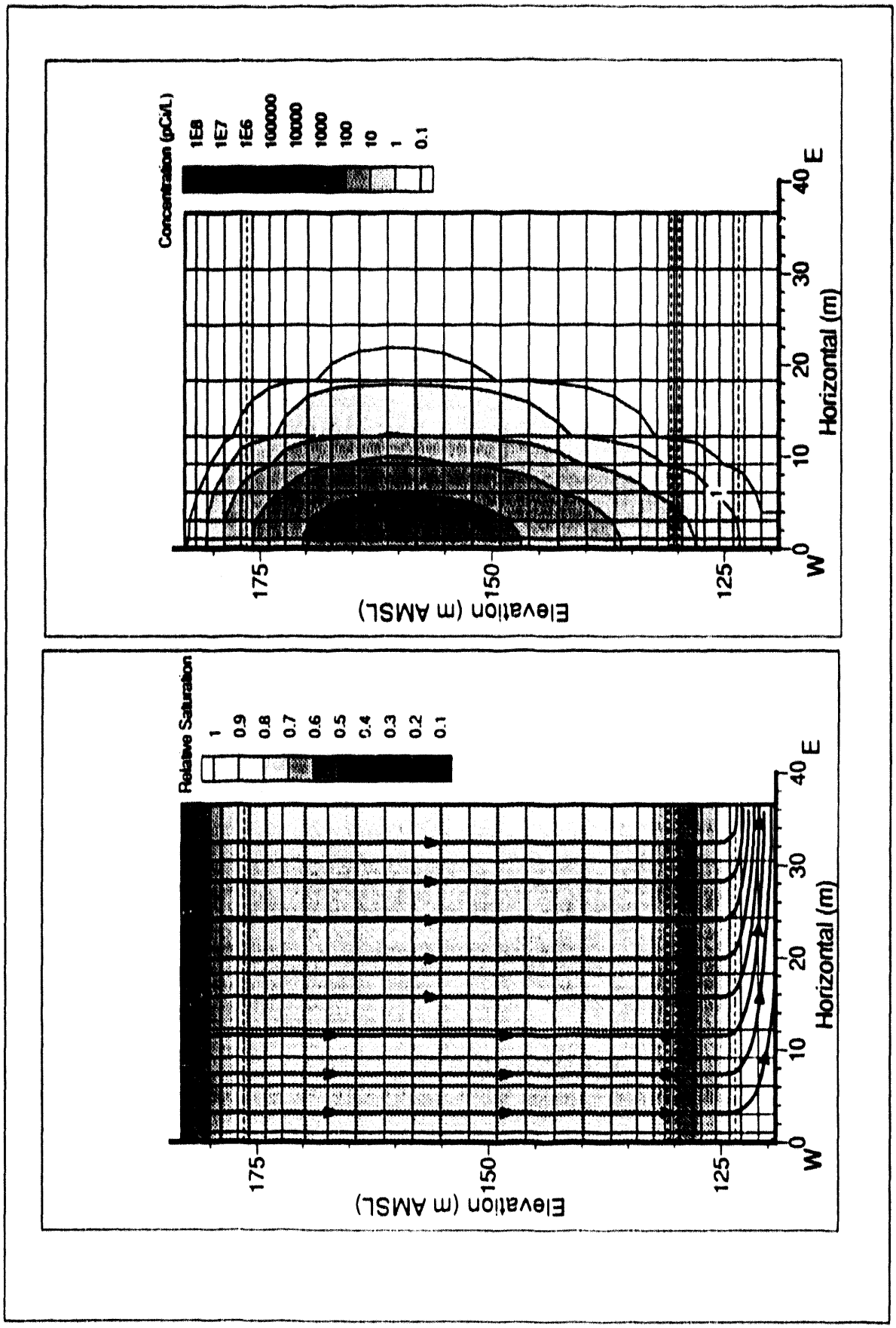
913-178402021/12-30-92

Figure 5-21. Relative Saturation, Streamlines, and Pu Concentration for Scenario A at Cessation of Discharge.



913-176/27022/12-30-92

Figure 5-22. Relative Saturation, Streamlines, and Pu Concentration for Scenario A after 150 yr.



913-1784/27023/12-30-92

Figure 5-23. Relative Saturation, Streamlines, and Pu Concentration for Scenario A after 9,500 yr.

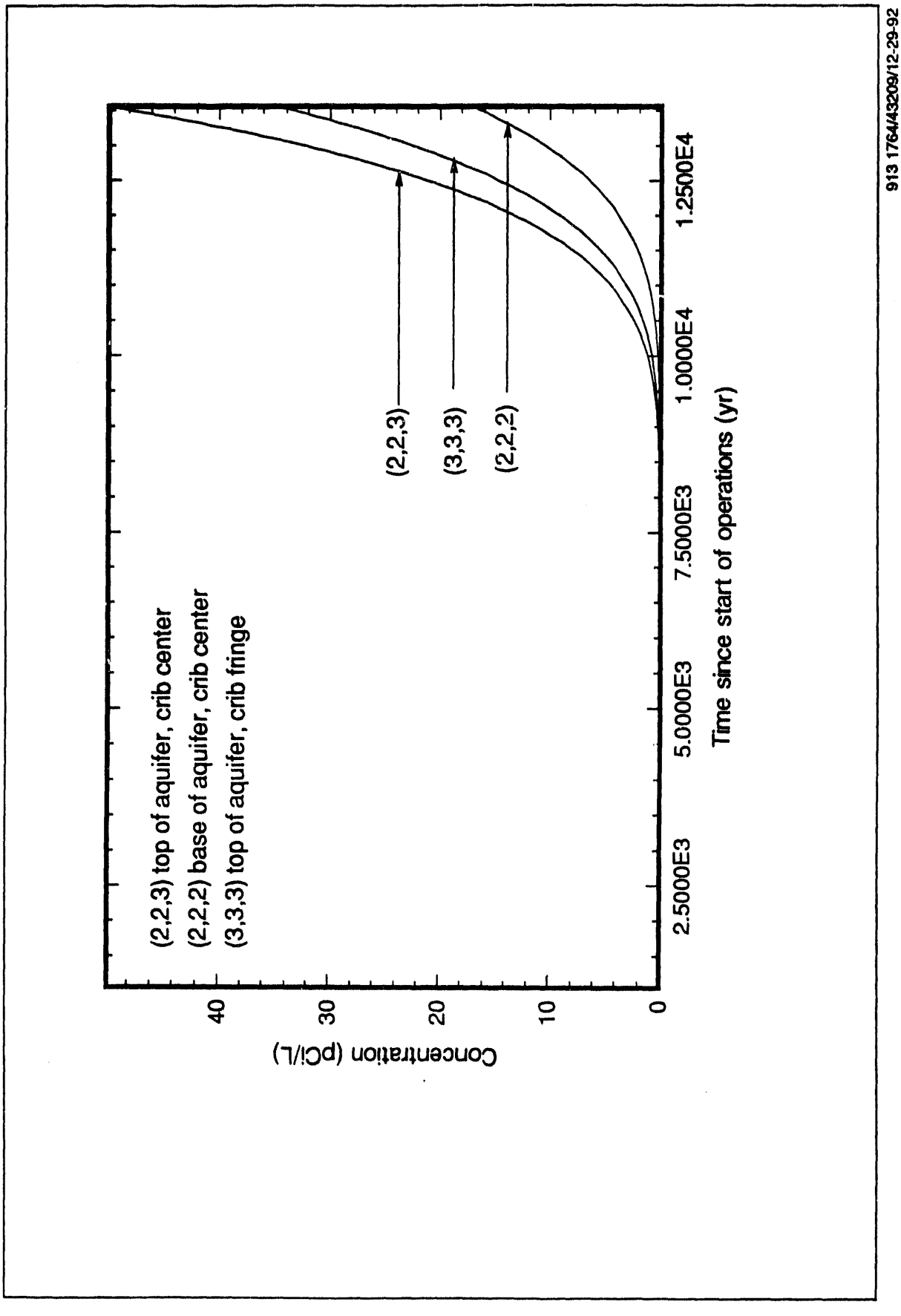


Figure 5-24. Concentration of Pu in the Unconfined Aquifer With Time for Scenario A.

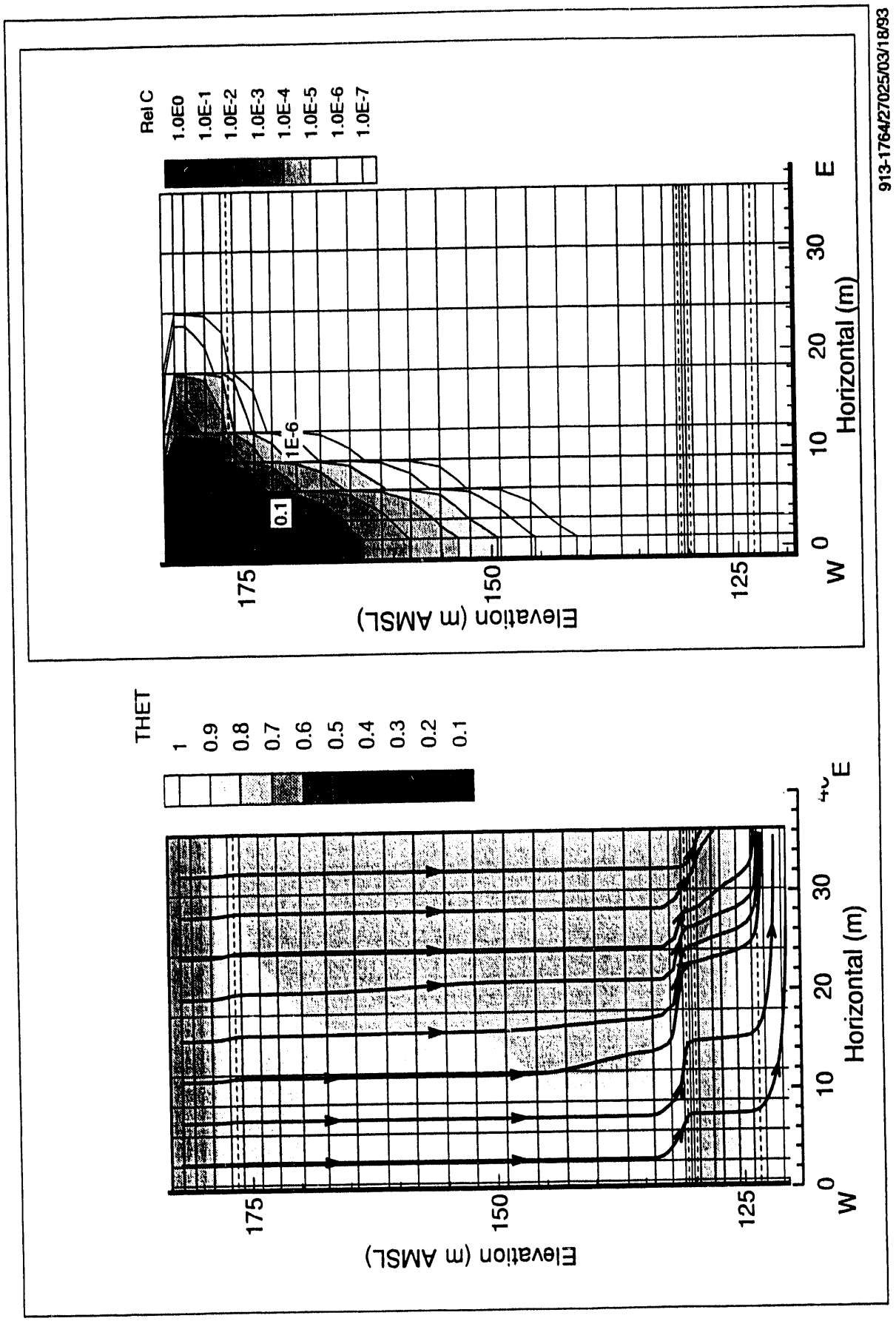
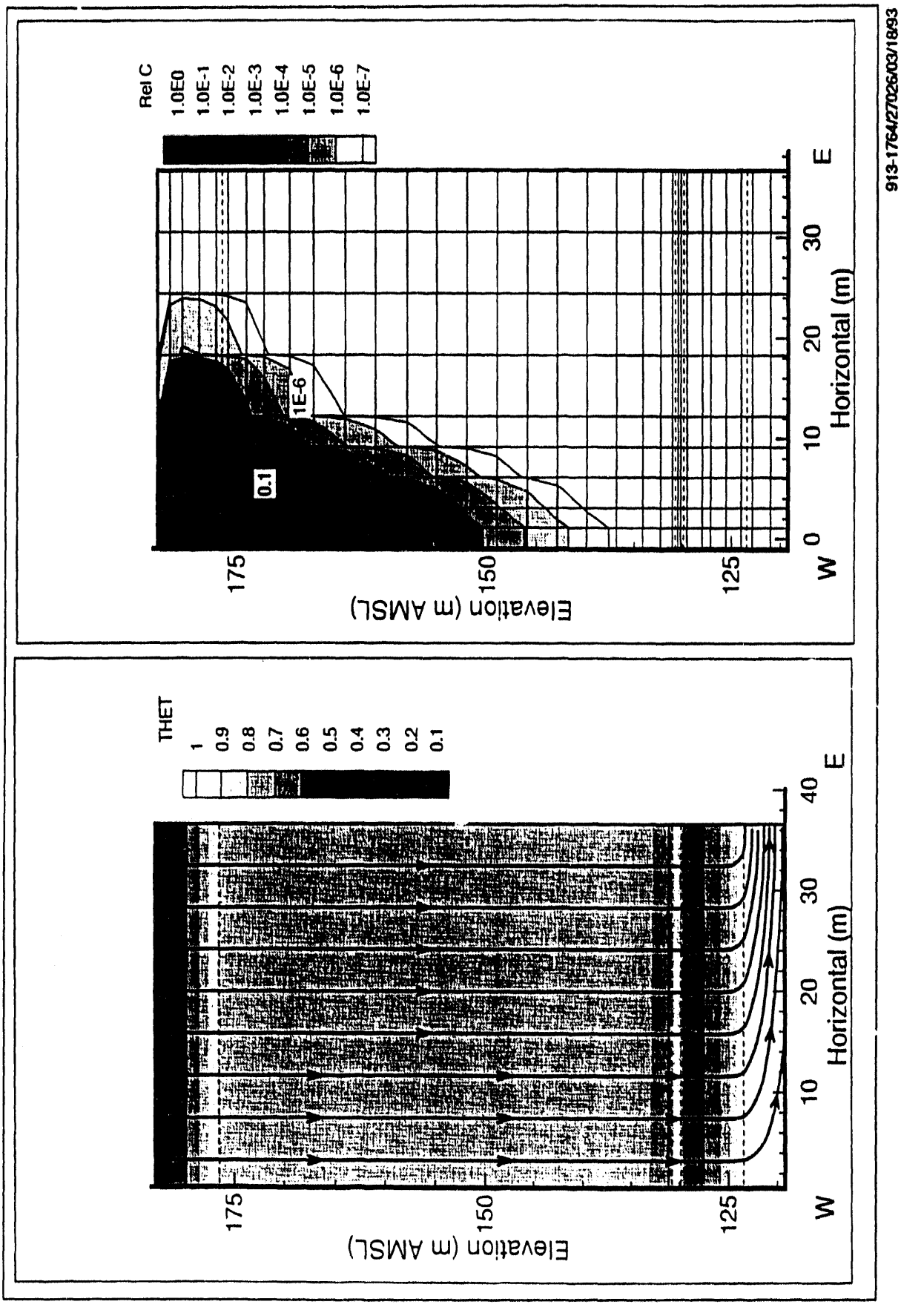


Figure 5-25. Relative Saturation, Streamlines, and TBP Concentration for Scenario A at Cessation of Discharge.



913-1764/27026/03/18/93

Figure 5-26. Relative Saturation, Streamlines, and TBP Concentration for Scenario A after 37 yr.

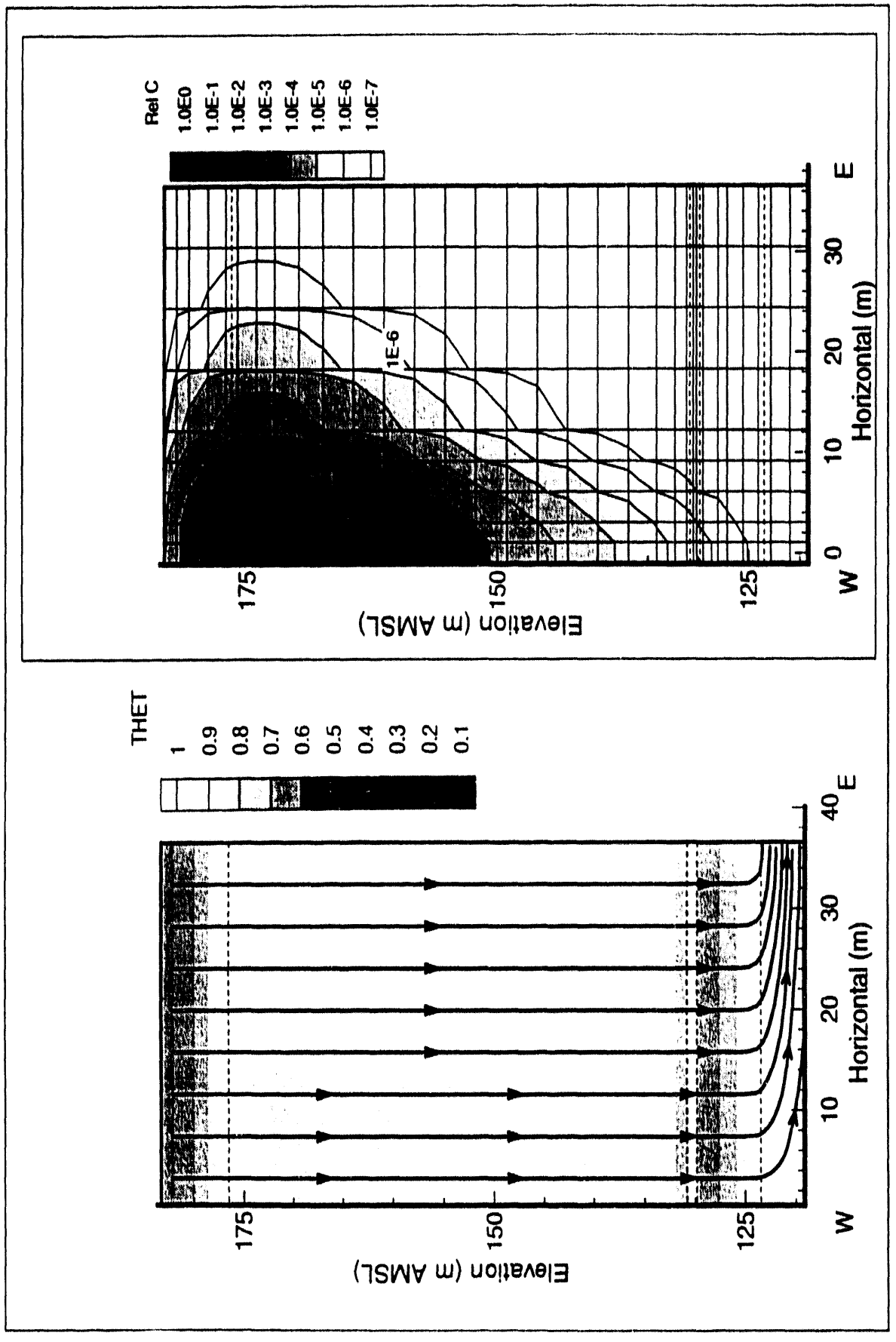
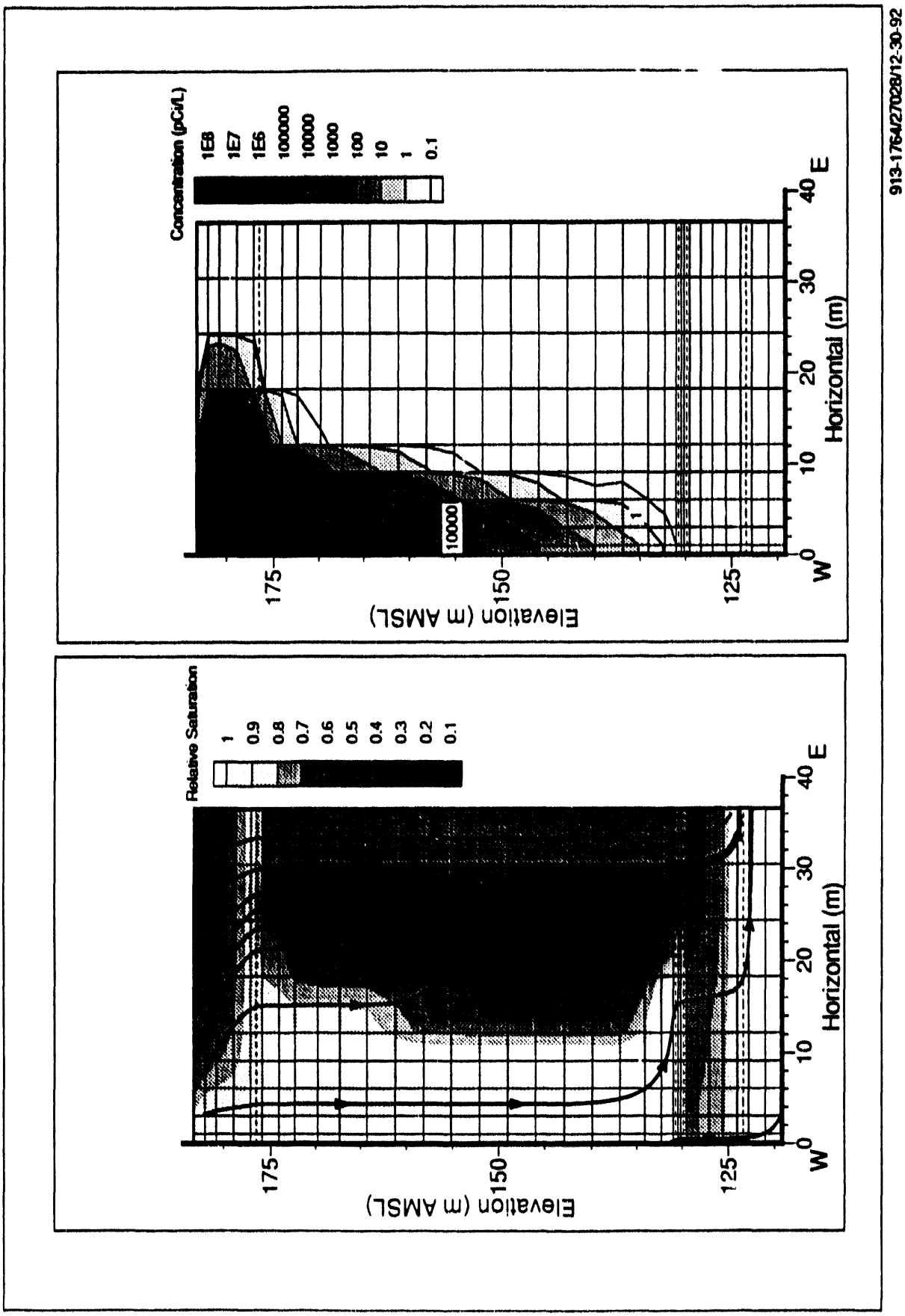
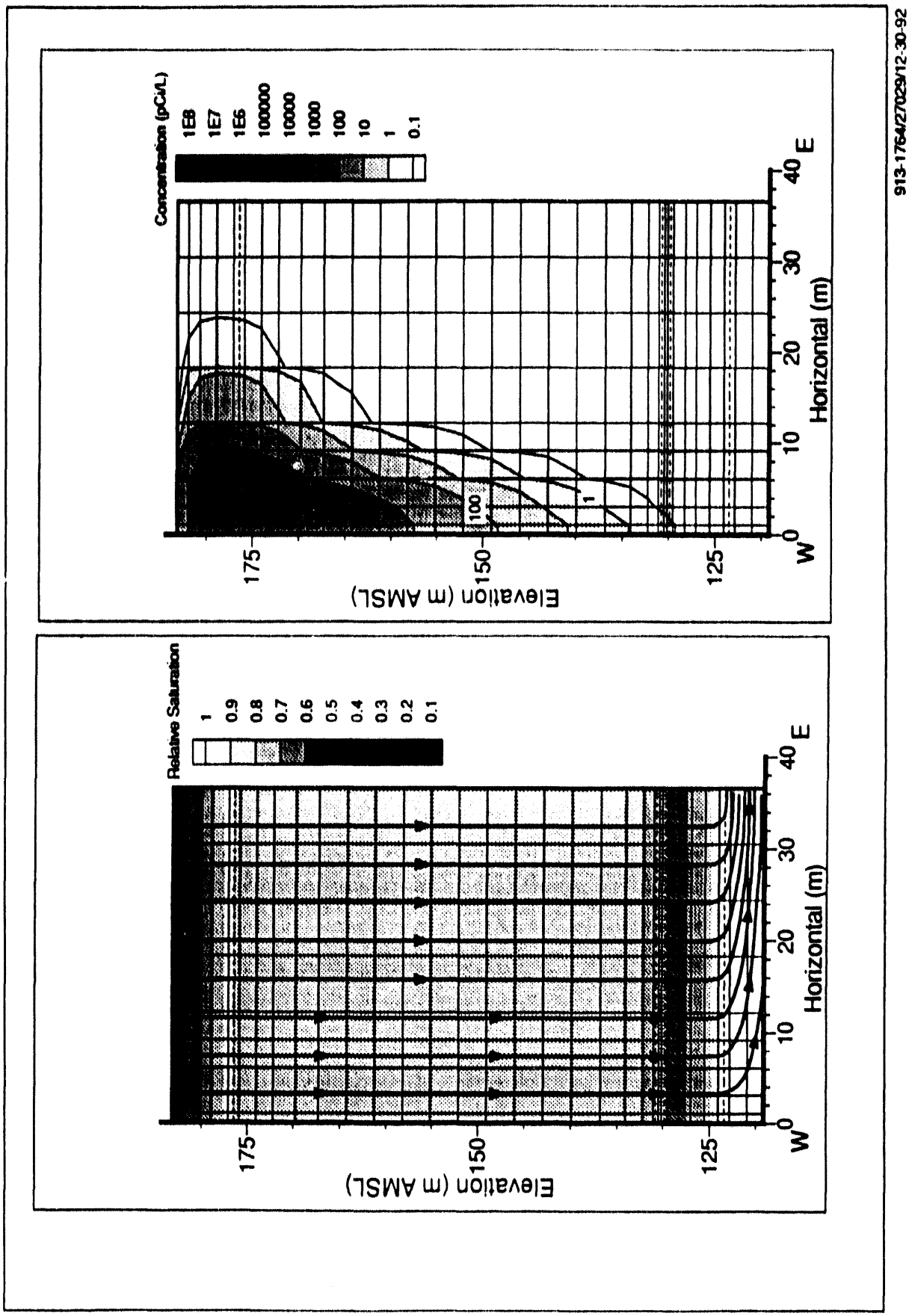


Figure 5-27. Relative Saturation, Streamlines, and TBP Concentration for Scenario A after 150 yr.



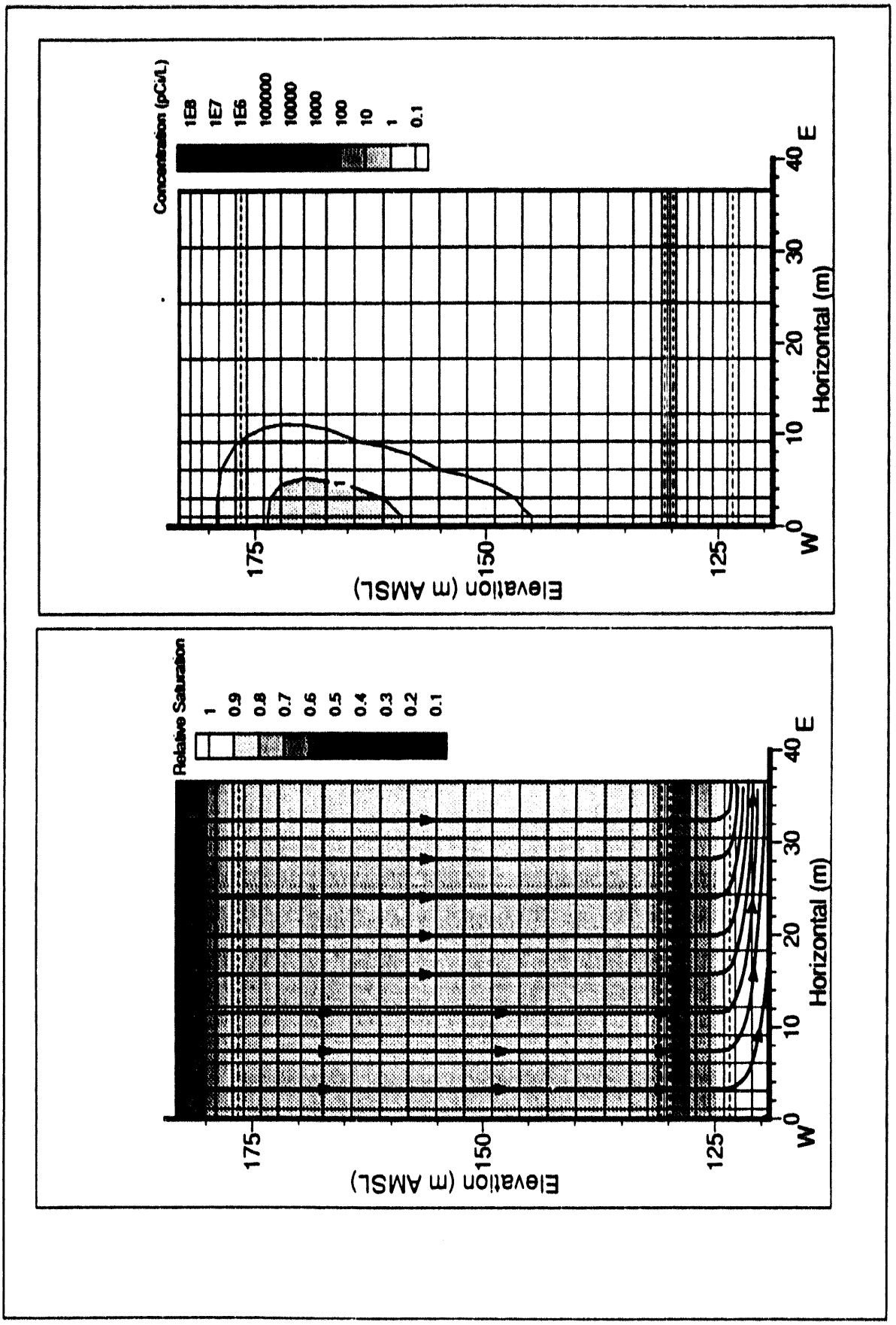
913-176/4/27028/12-30-92

Figure 5-28. Relative Saturation, Streamlines, and Co-60 Concentration for Scenario A at Cessation of Discharge.



913-1764/27029/12-30-92

Figure 5-29. Relative Saturation, Streamlines, and Co-60 Concentration for Scenario A after 37 yr.



913-176\27030\12-30-92

Figure 5-30. Relative Saturation, Streamlines, and Co-60 Concentration for Scenario A after 100 yr.

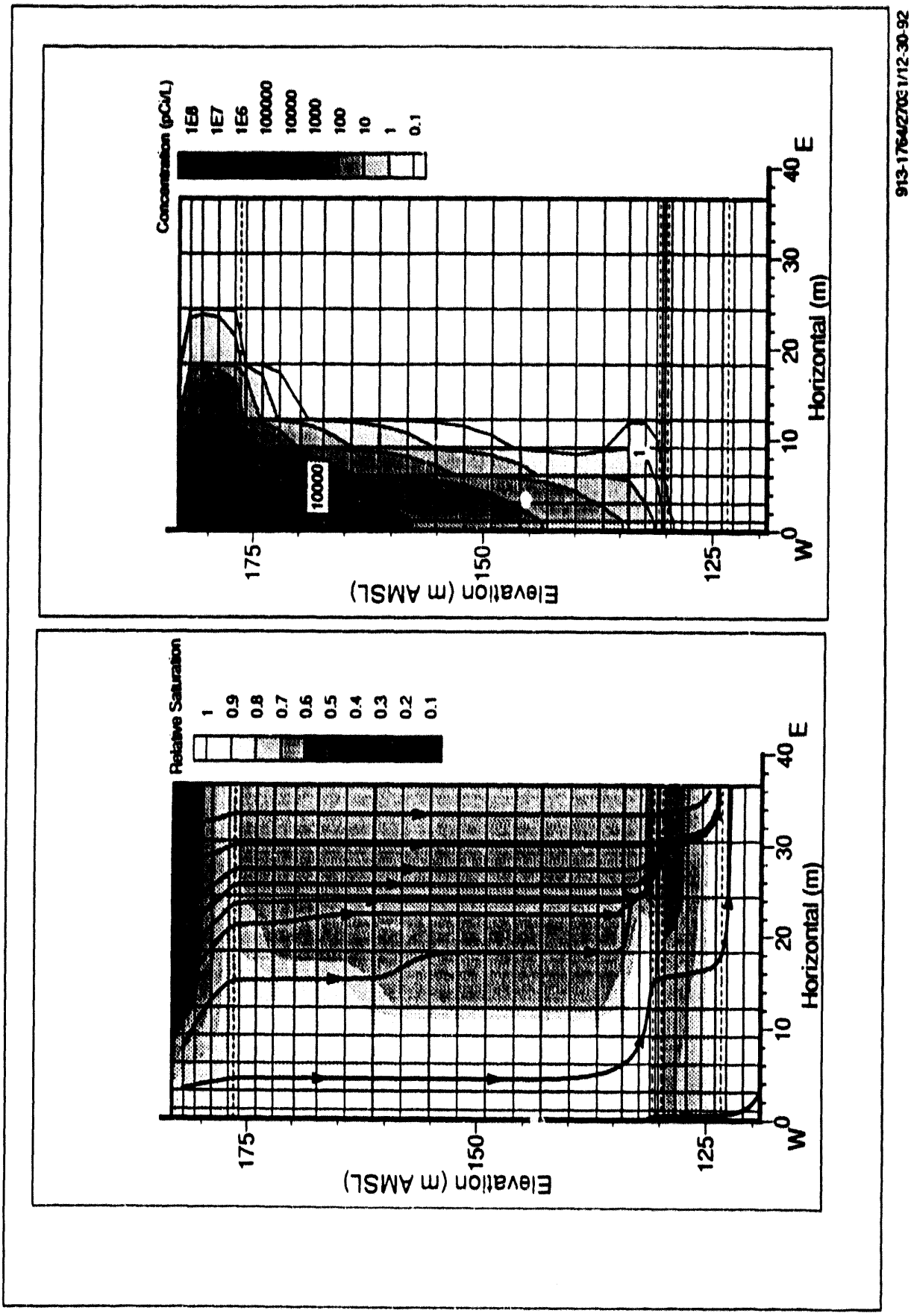
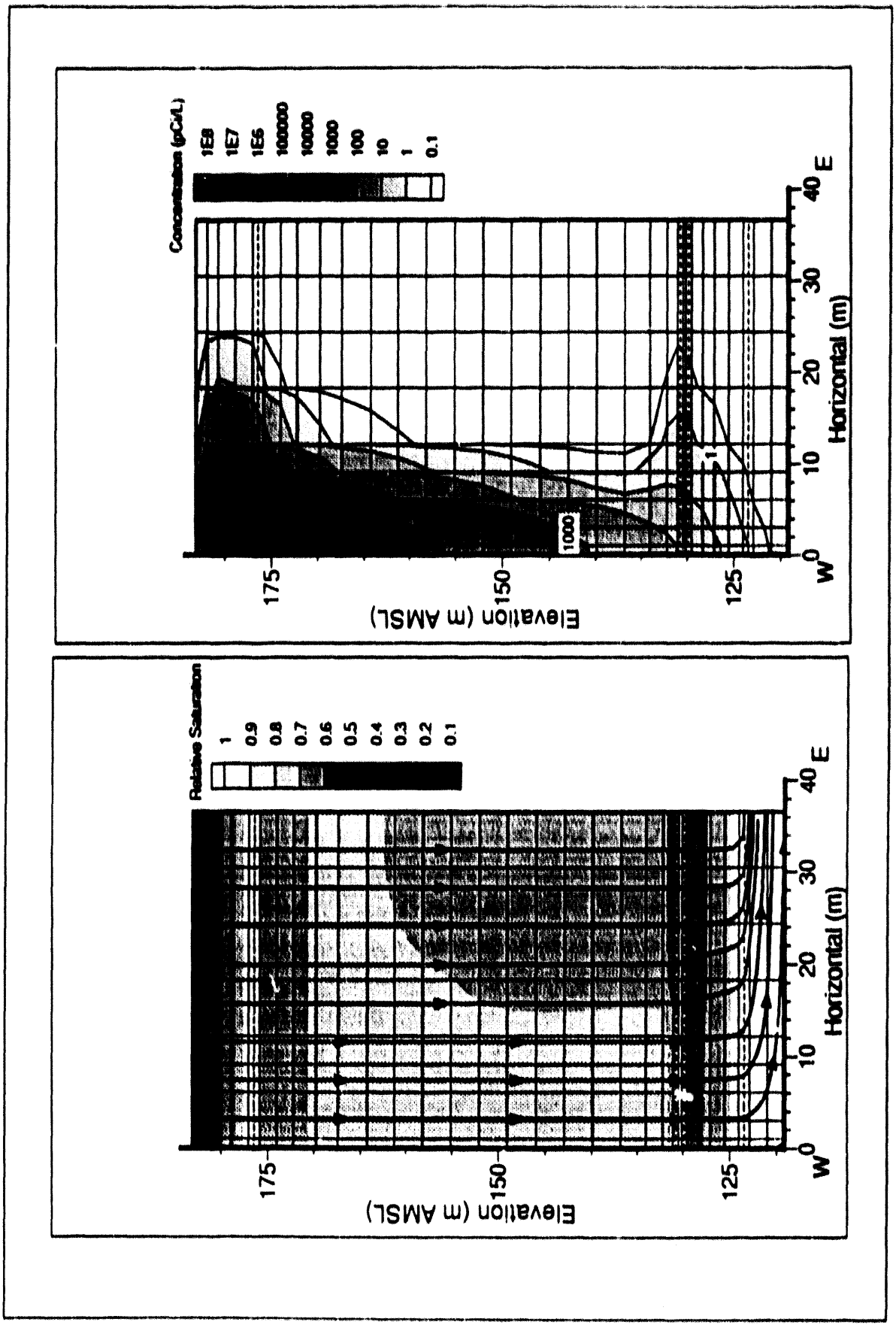
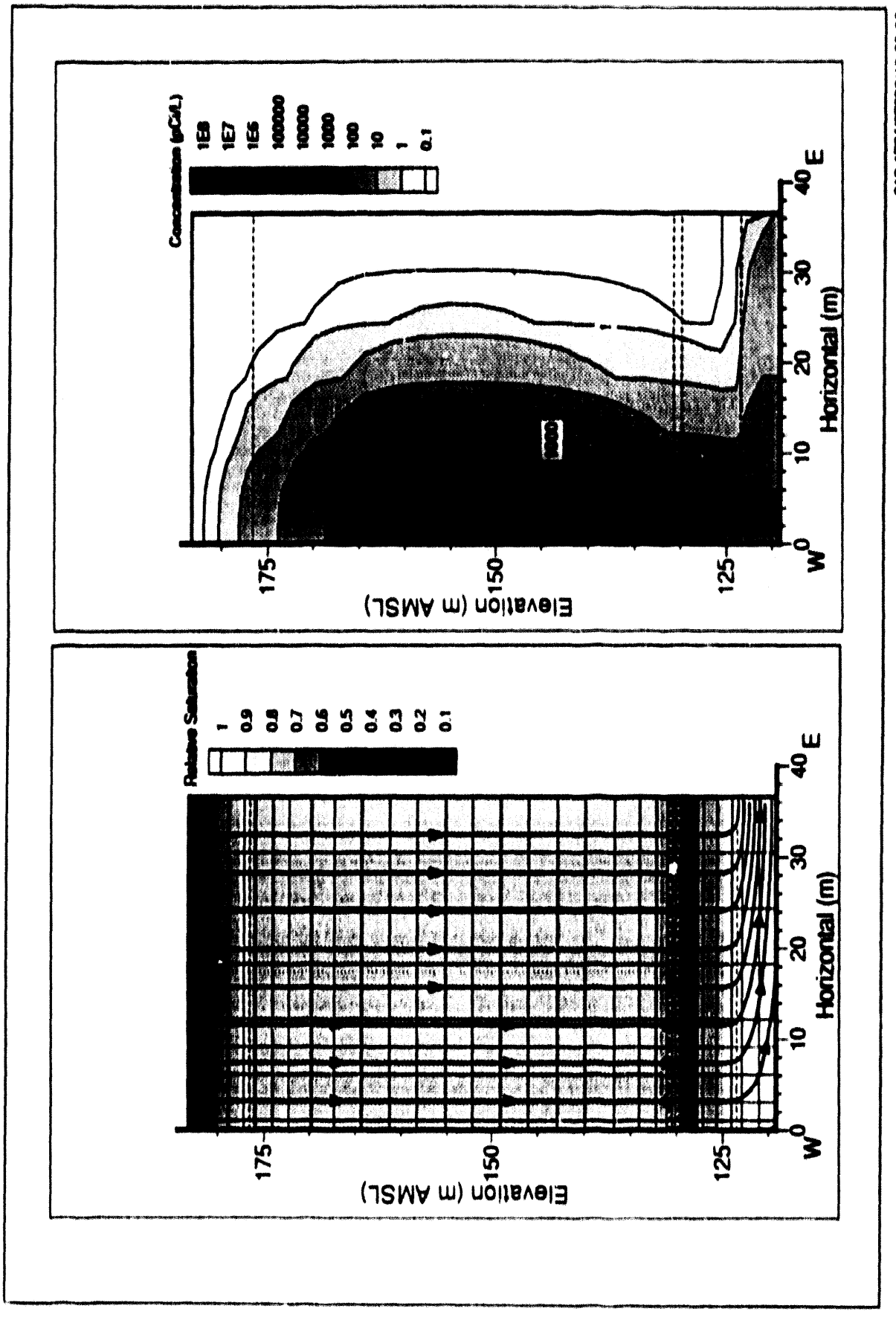


Figure 5-31. Relative Saturation, Streamlines, and Total U Concentration for Scenario A at Cessation of Discharge.



913-176/27032/12-30-92

Figure 5-32. Relative Saturation, Streamlines, and Total U Concentration for Scenario A after 3 yr.



913-1764/7033/12-30-92

Figure 5-33. Relative Saturation, Streamlines, and Total U Concentration for Scenario A after 150 yr.

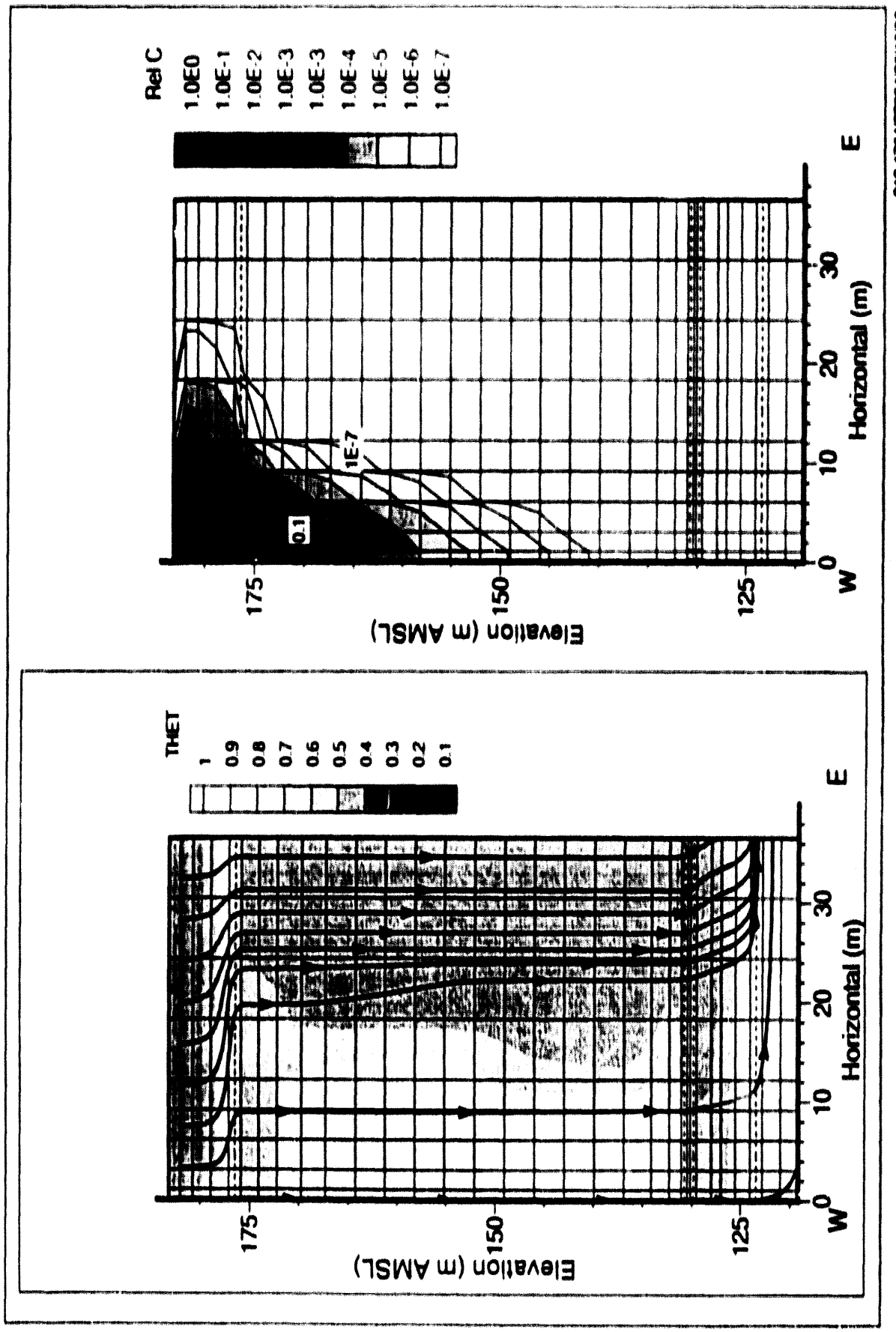
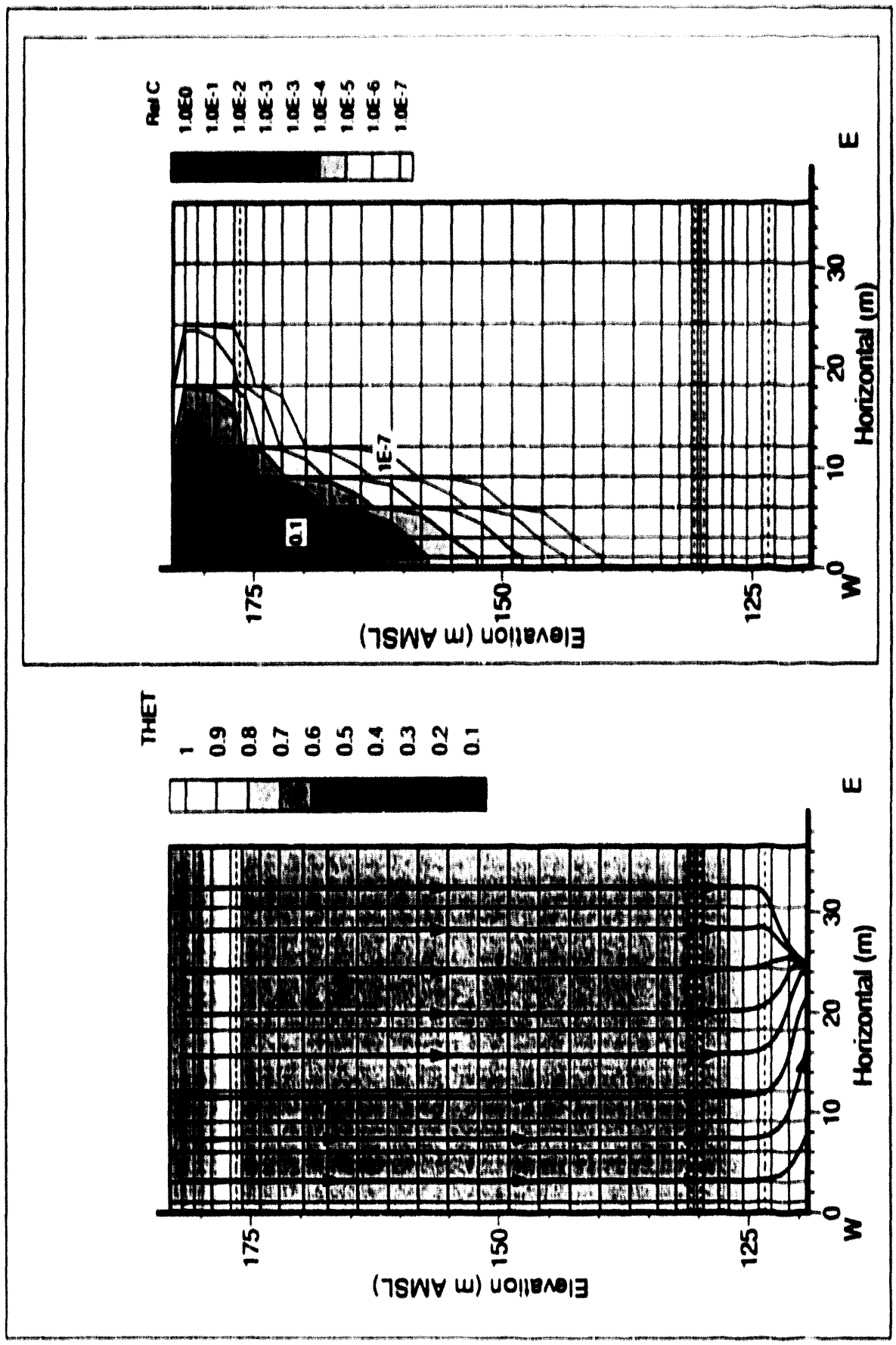


Figure 5-34. Relative Saturation, Streamlines, and TBP Concentration for Scenario B at Cessation of Discharge.



913-176427035031893

Figure 5-35. Relative Saturation, Streamlines, and TBP Concentration for Scenario B after 37 yr.

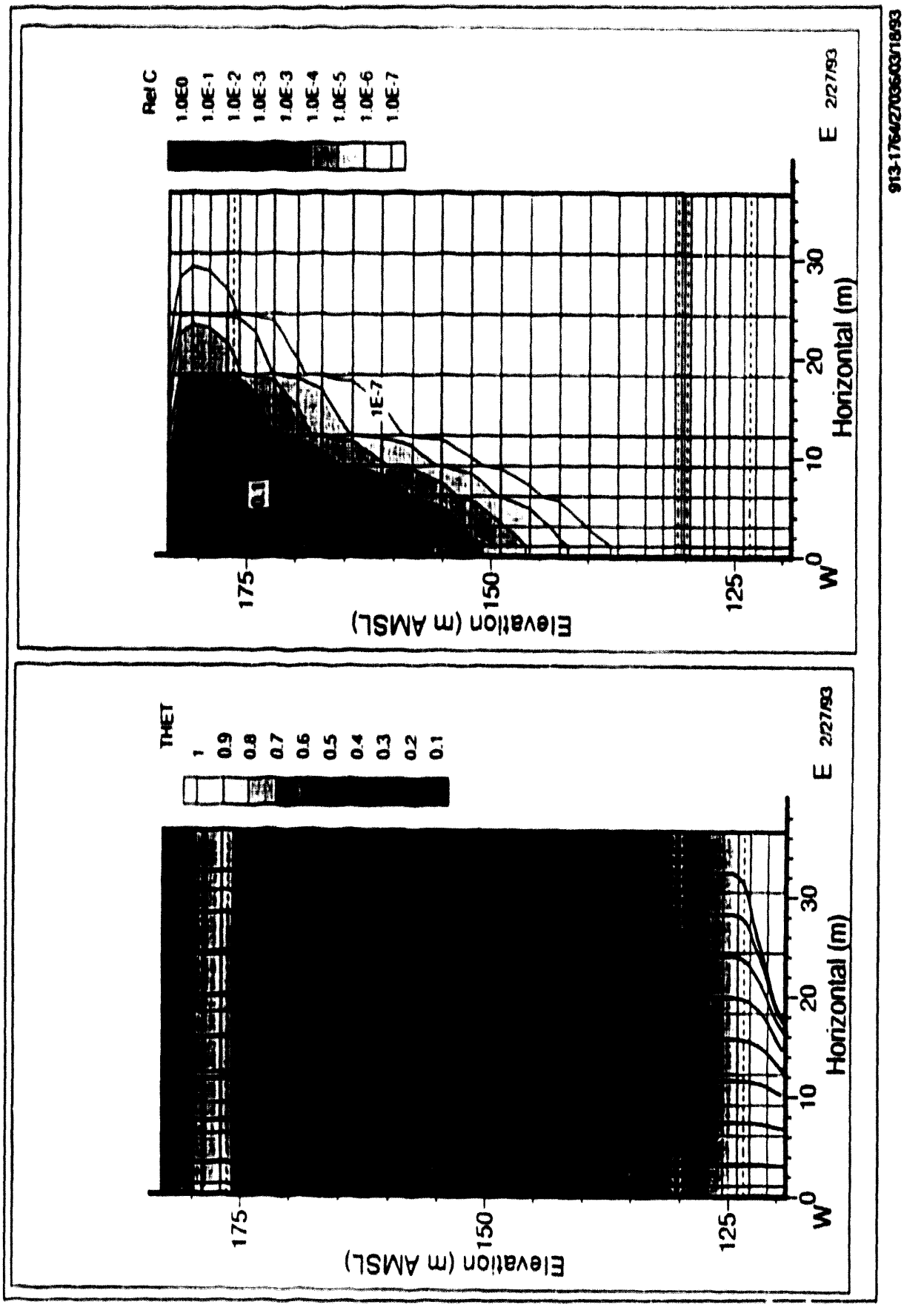


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Figure 5-36. Relative Saturation, Streamlines, and TBP Concentration for Scenario B after 300 yr.

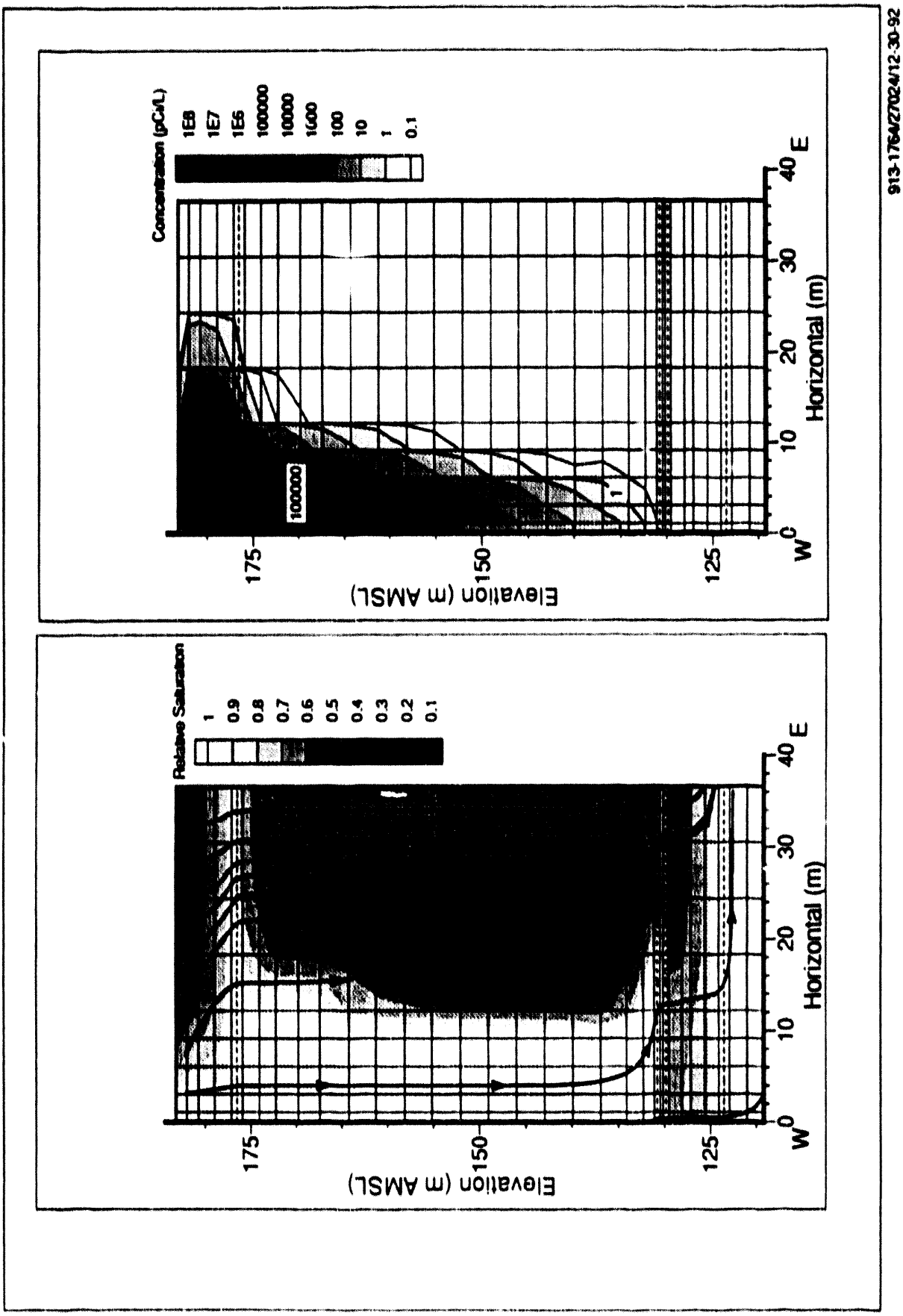

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Figure 5-37. Relative Saturation, Streamlines, and Co-60 Concentration for Scenario B at Cessation of Discharge.

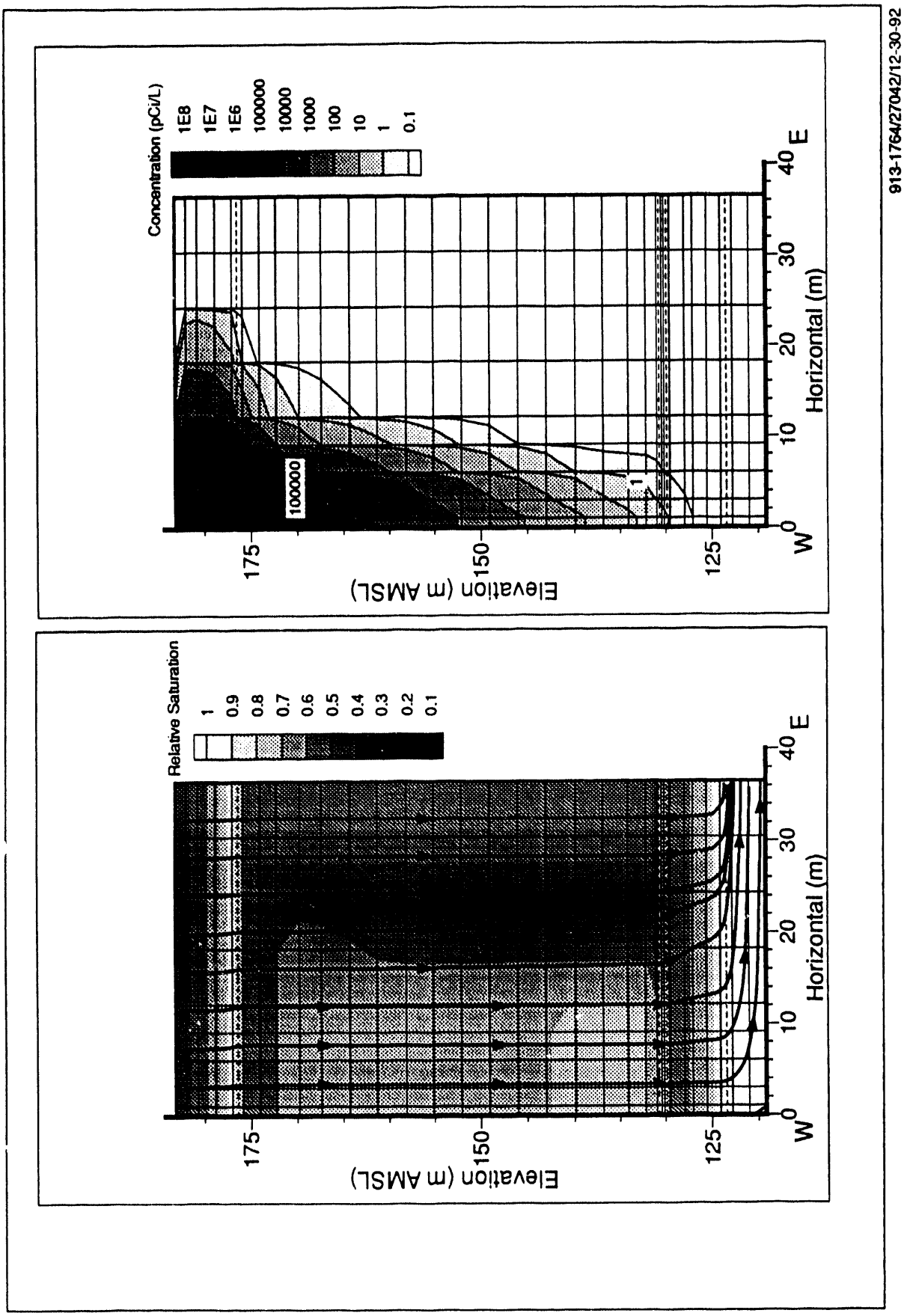

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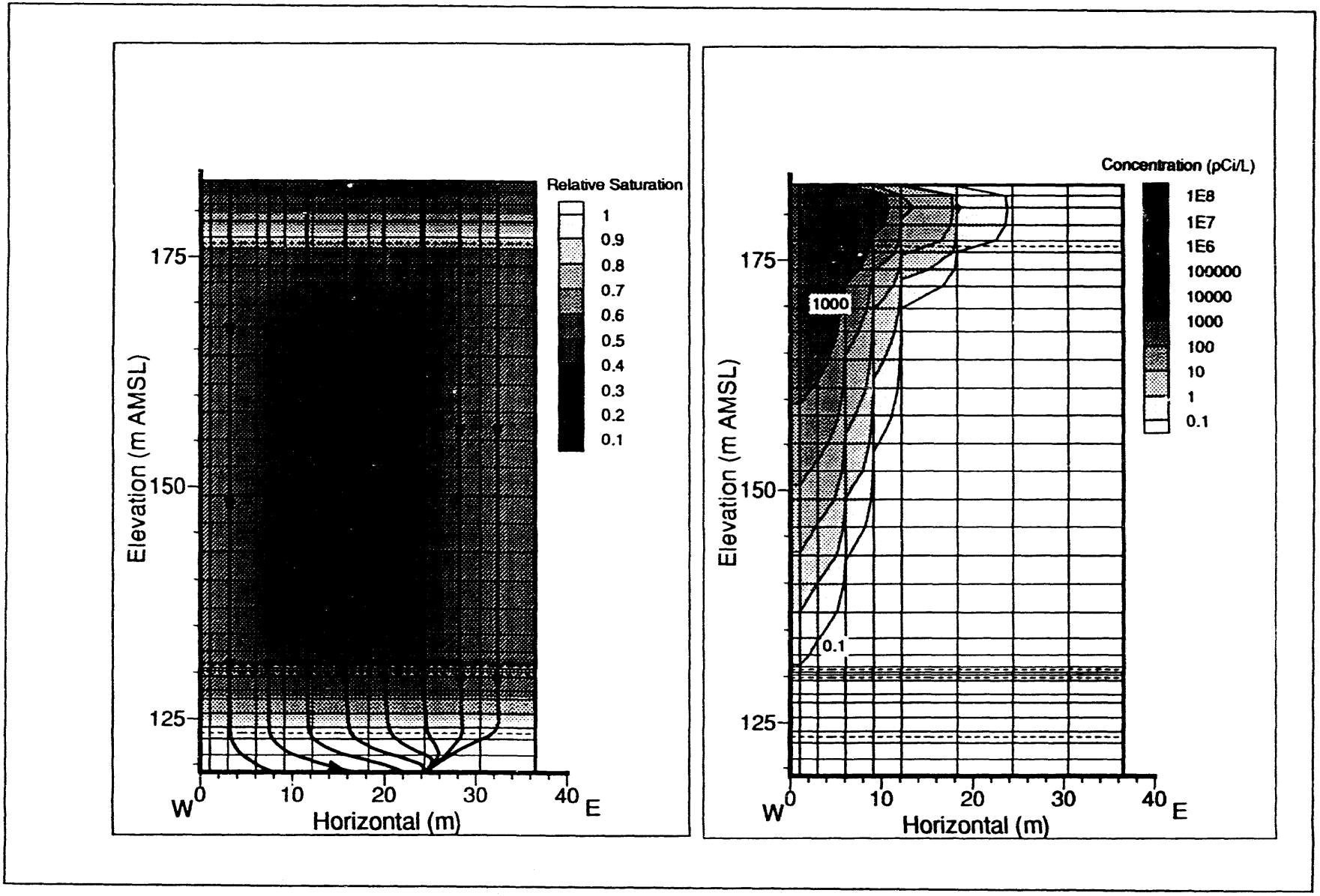
Figure 5-38. Relative Saturation, Streamlines, and Co-60 Concentration for Scenario B after 5 yr.



913-1764/27043/12-30-92

Figure 5-39. Relative Saturation, Streamlines, and Co-60 Concentration for Scenario B after 37 yr.

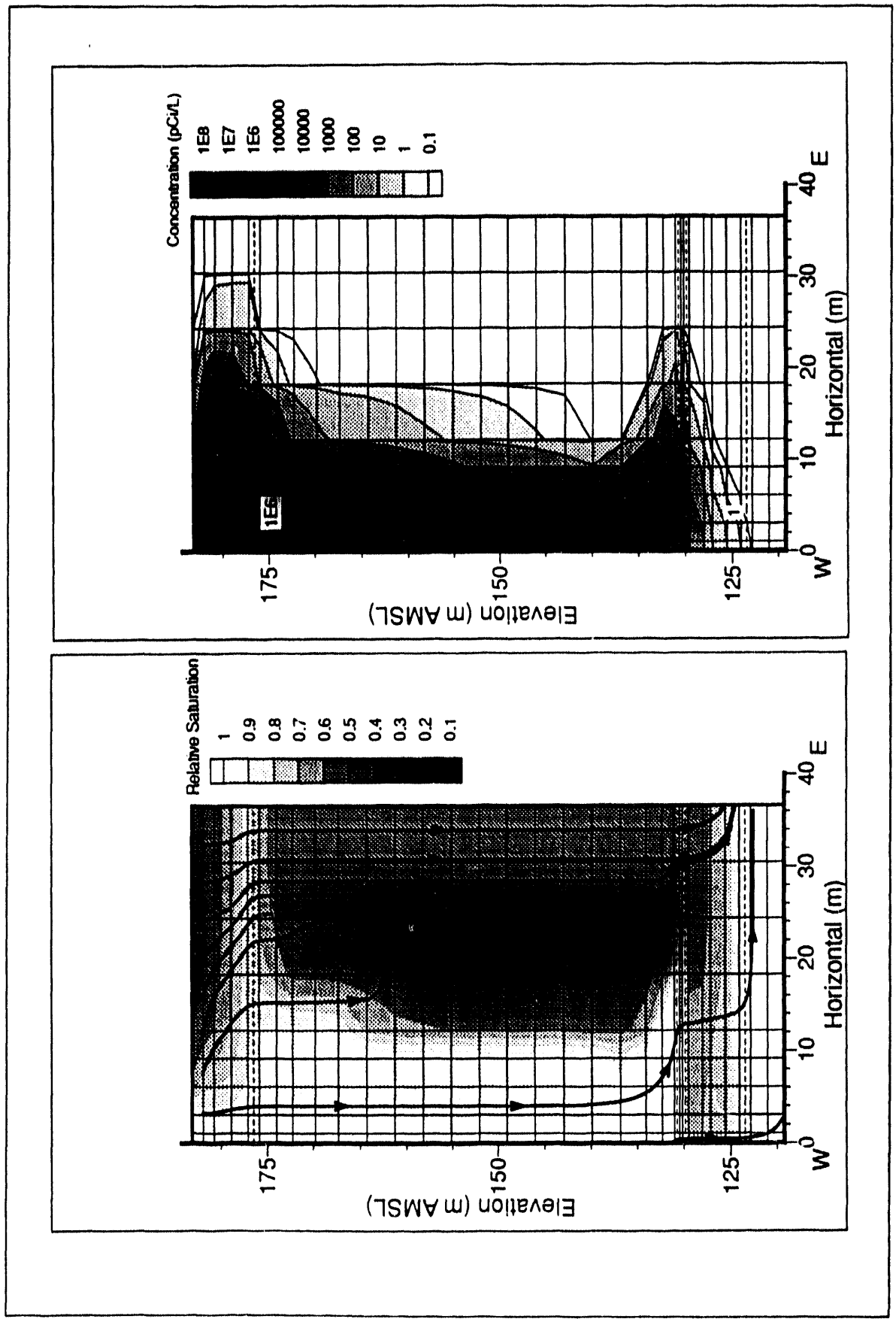
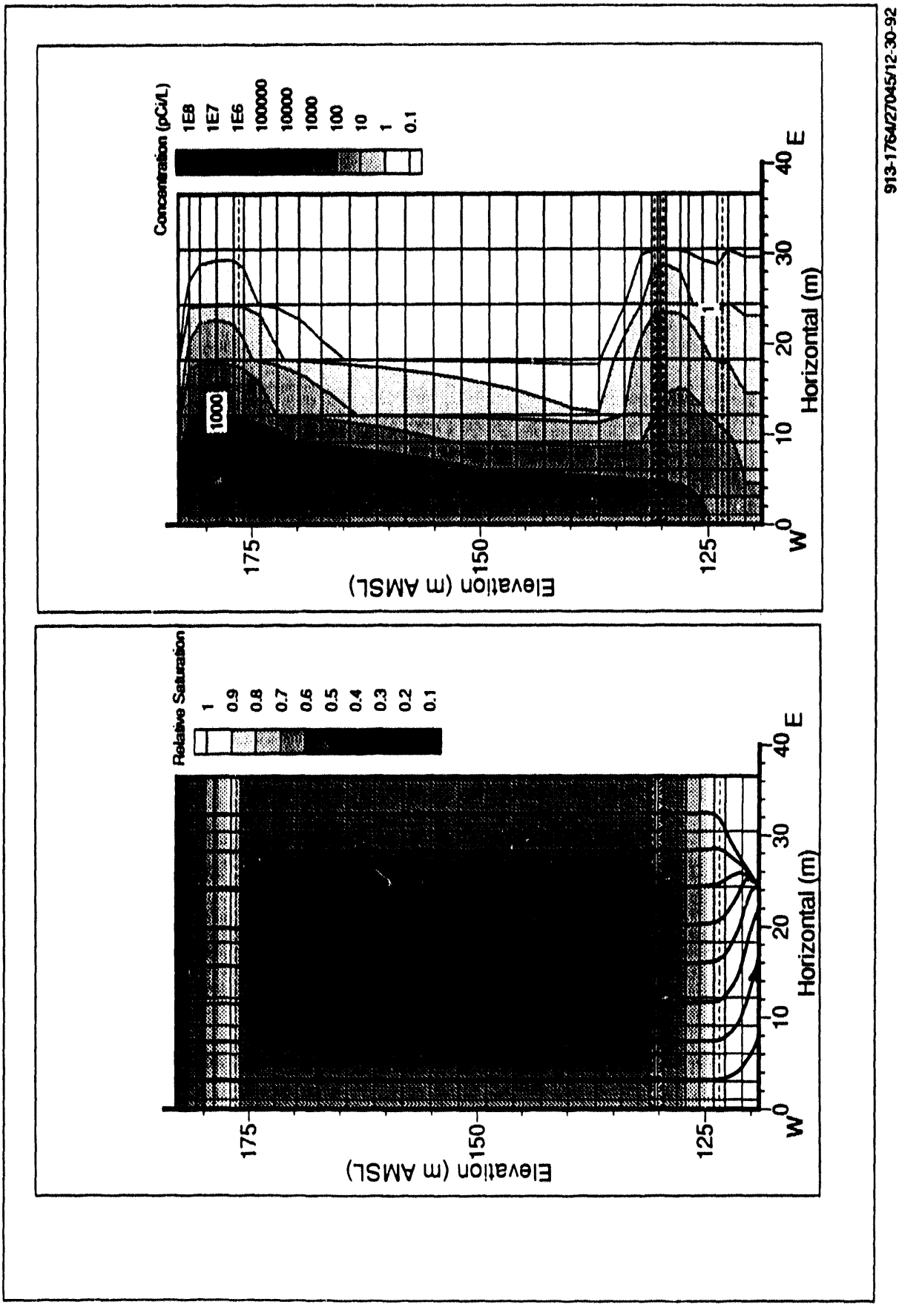
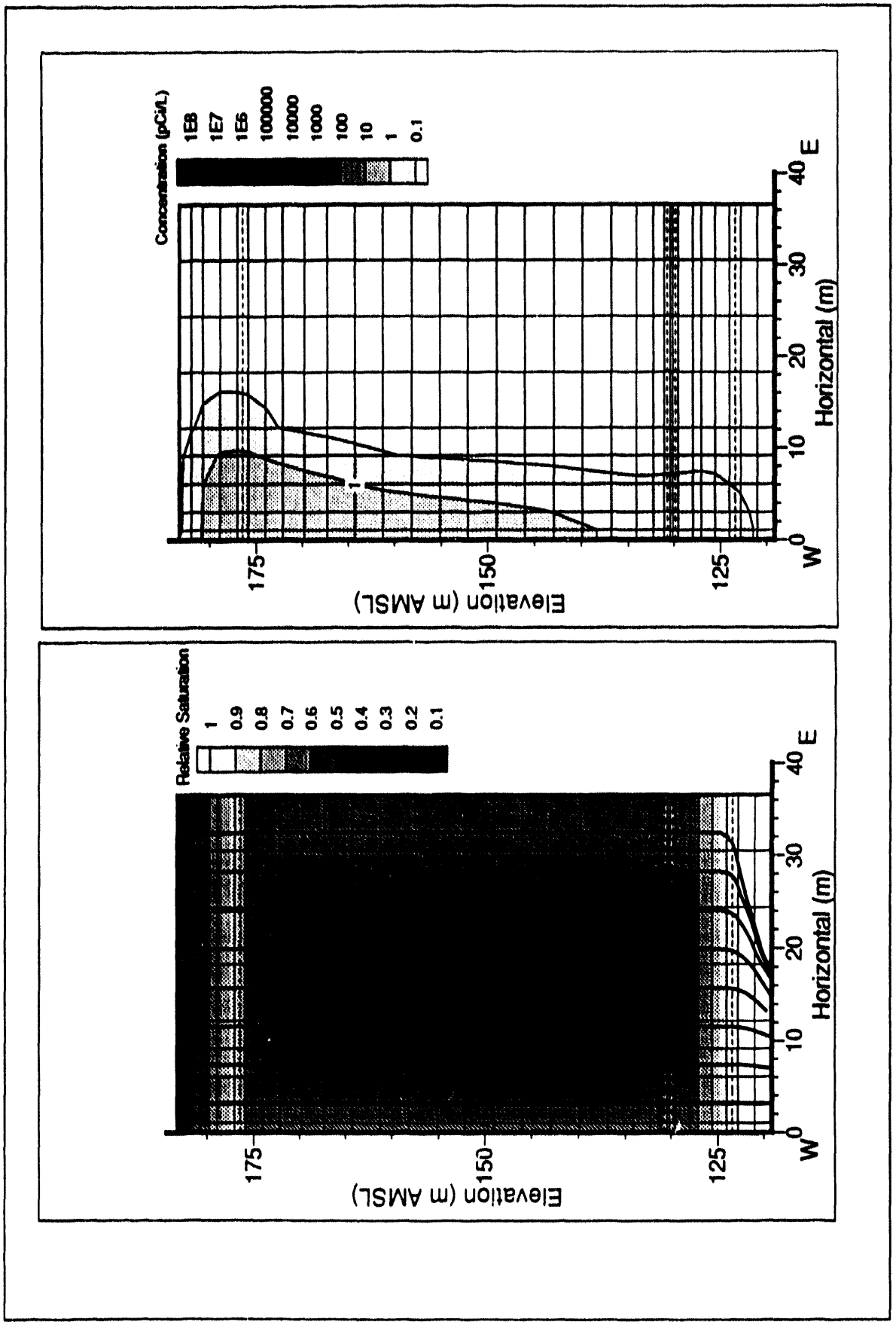


Figure 5-40. Relative Saturation, Streamlines, and Co-60 Concentration Using a Distribution Coefficient of 0.5 mL/g for Scenario B at Cessation of Discharge.



913-1764/27045/12-30-92

Figure 5-41. Relative Saturation, Streamlines, and Co-60 Concentration Using a Distribution Coefficient of 0.5 mL/g for Scenario B after 37 yr.



913-1764/27046/12-30-92

Figure 5-42. Relative Saturation, Streamlines, and Co-60 Concentration Using a Distribution Coefficient of 0.5 mL/g for Scenario B after 100 yr.

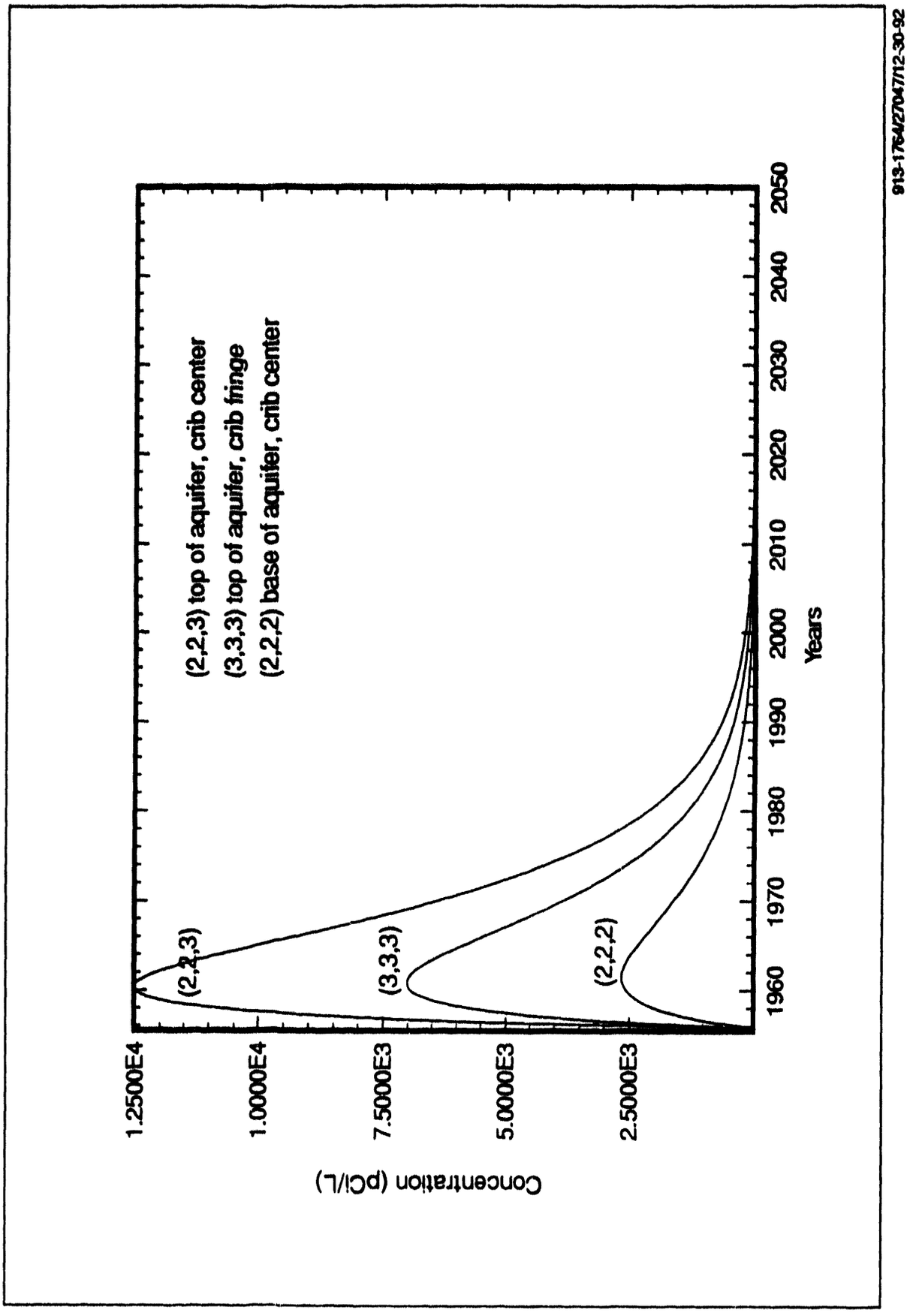
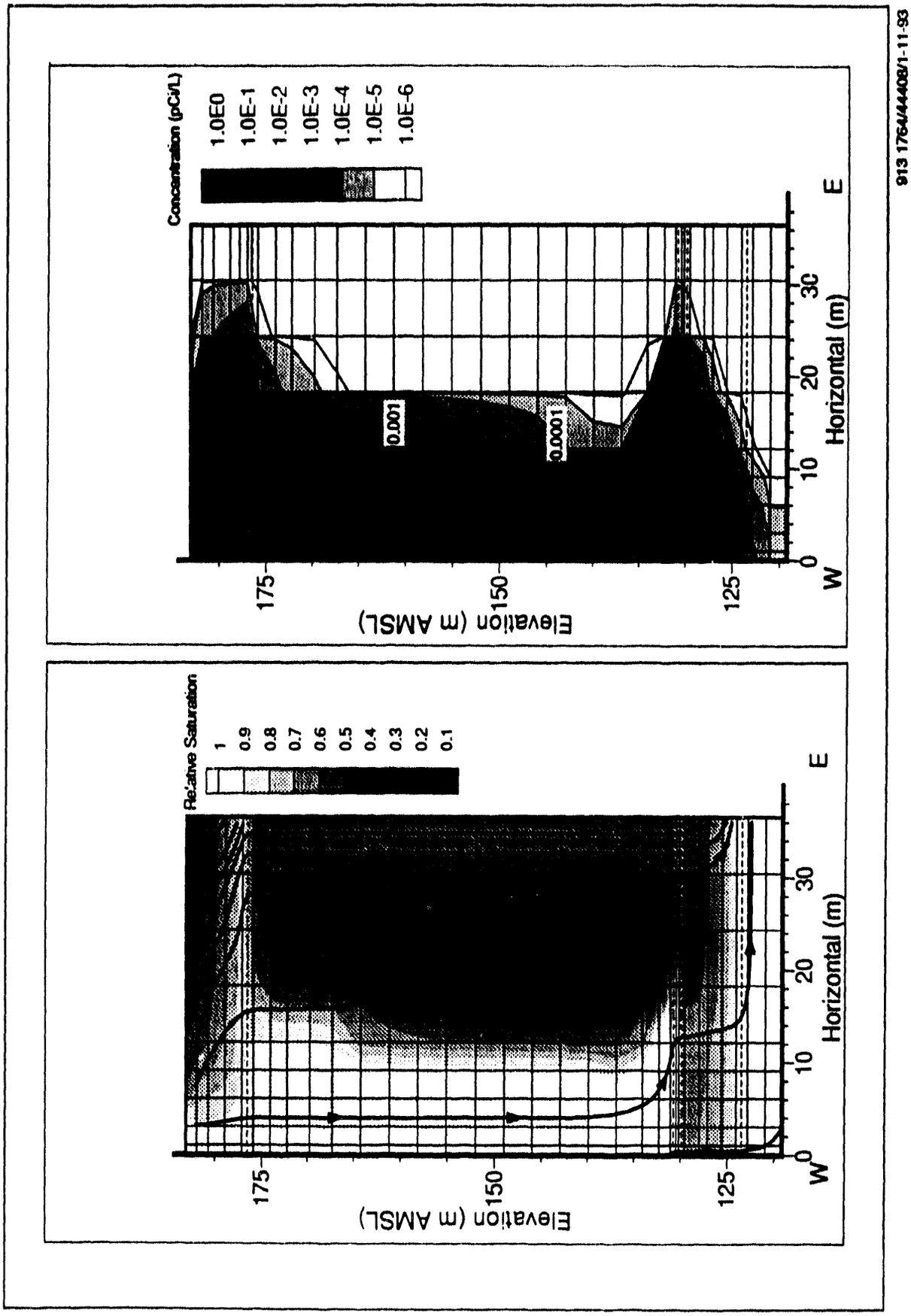


Figure 5-43. Concentration of Co-60 Using a Distribution Coefficient of 0.5 mL/g in the Unconfined Aquifer With Time for Scenario B.



913 1764/44408/1-11-93

Figure 5-44. Relative Saturation, Streamlines and Relative Tc-99 Concentration for Scenario B at Cessation of Discharge.

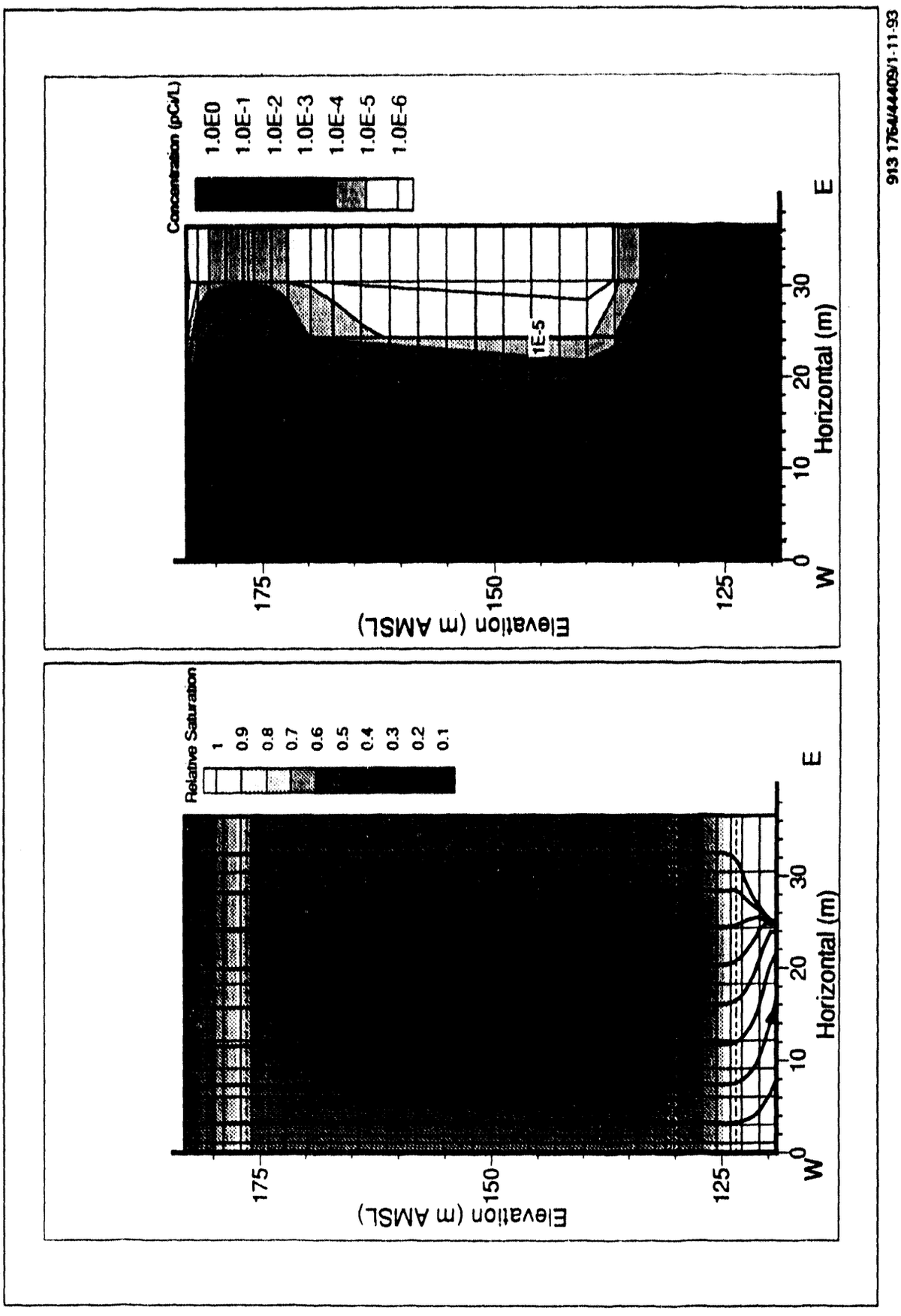
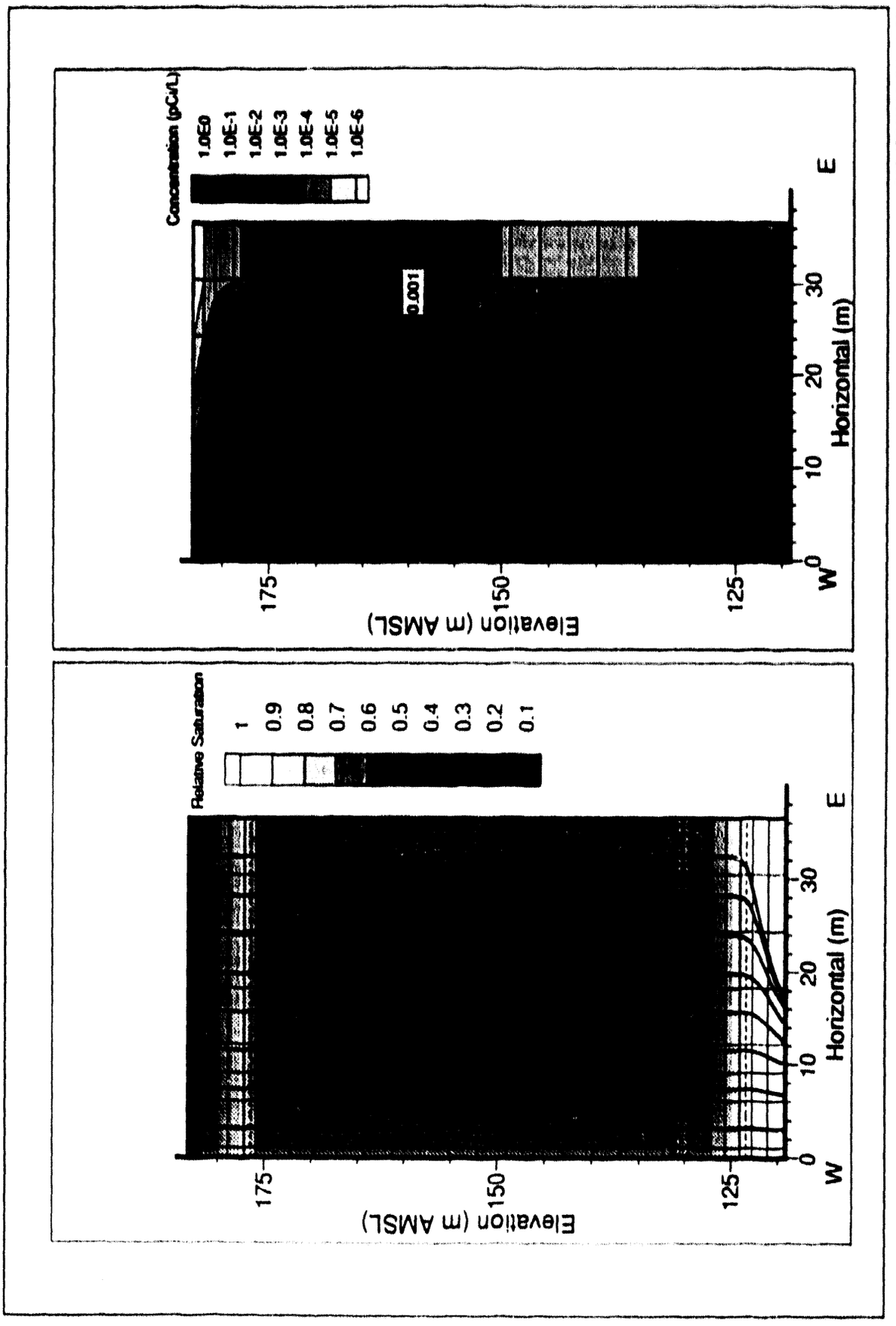
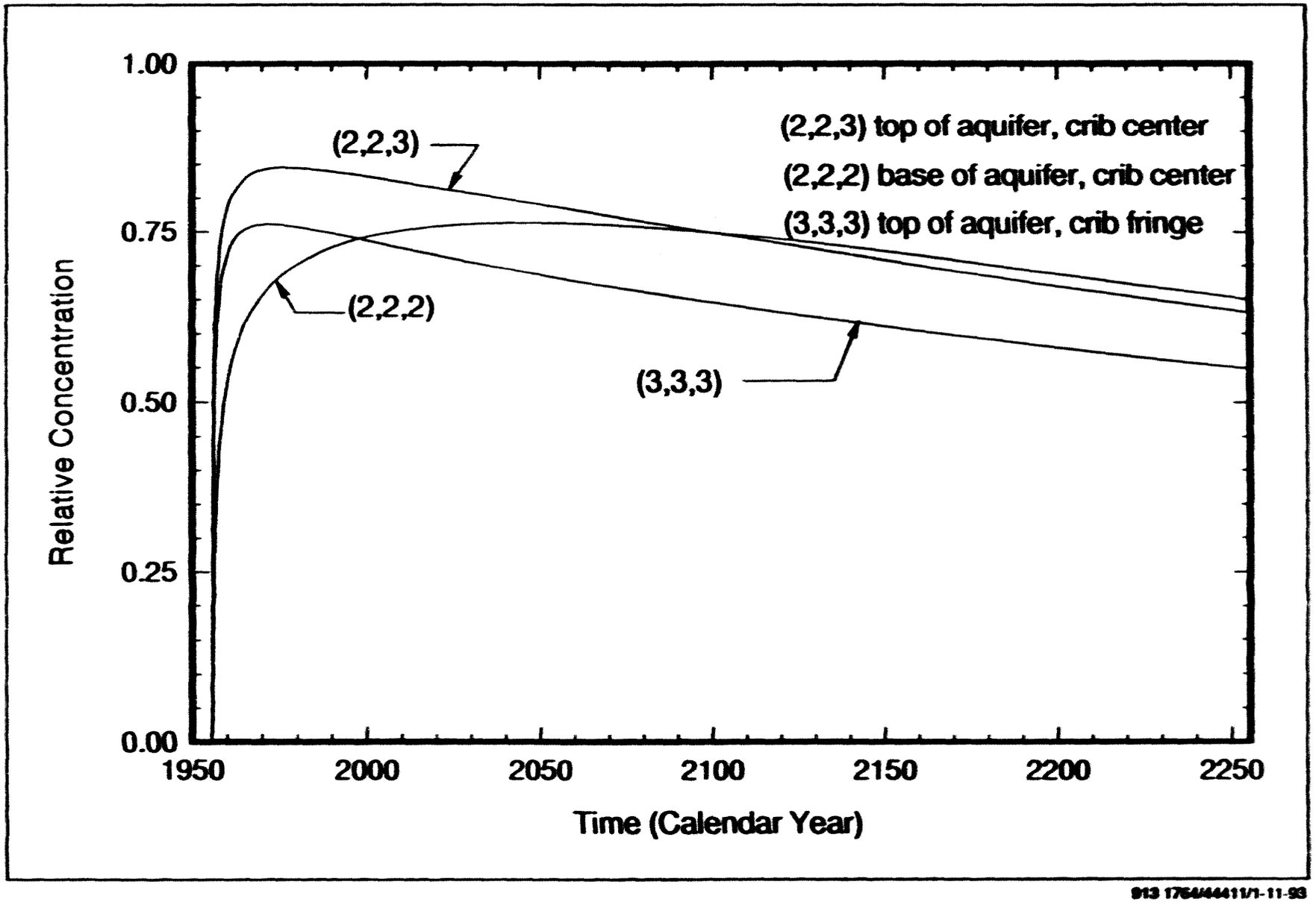


Figure 5-45. Relative Saturation, Streamlines and Relative Tc-99 Concentration for Scenario B After 37 Years.



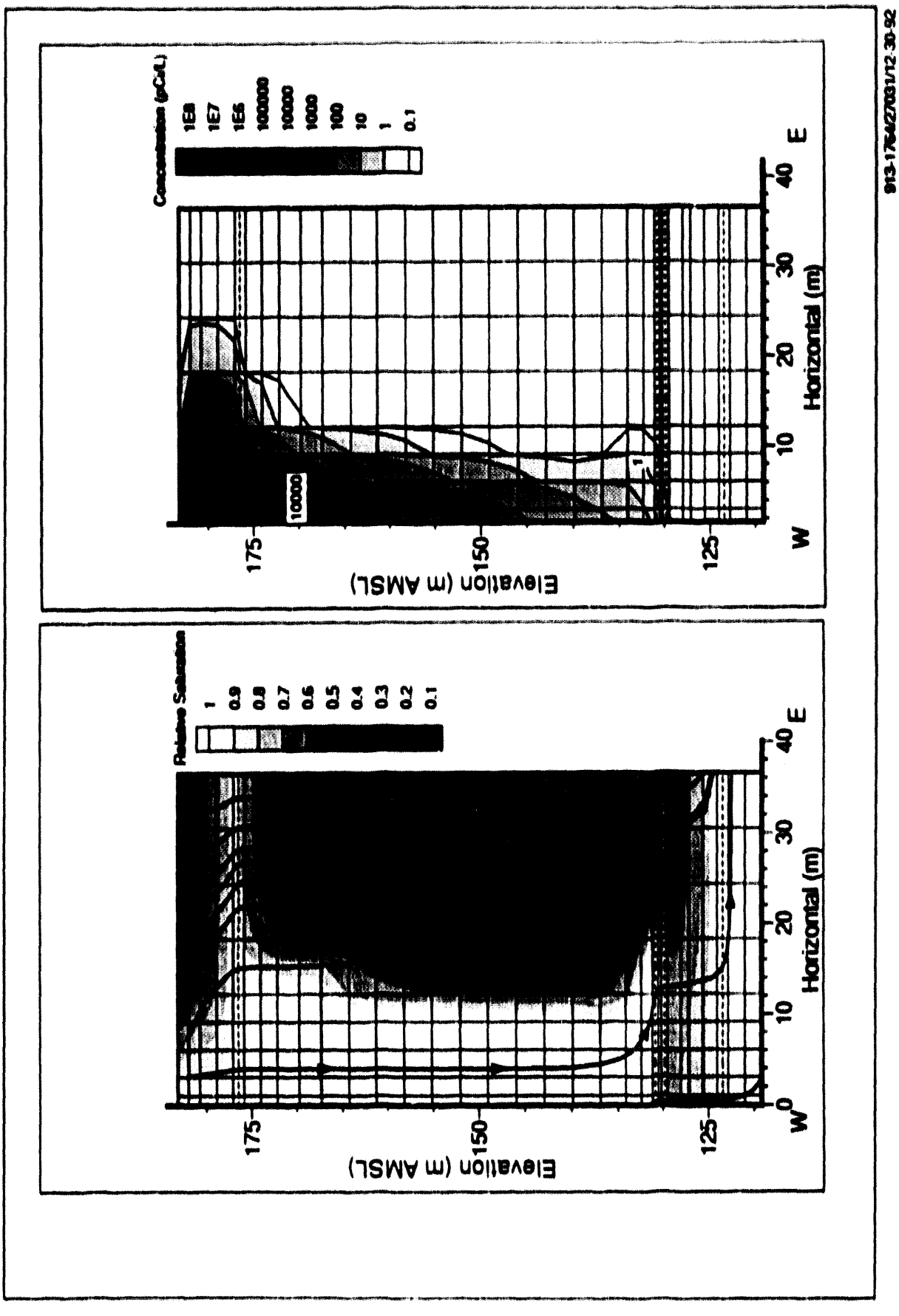
913 1764444101-11-93

Figure 5-46. Relative Saturation, Streamlines and Relative Tc-99 Concentration for Scenario B After 300 Years.



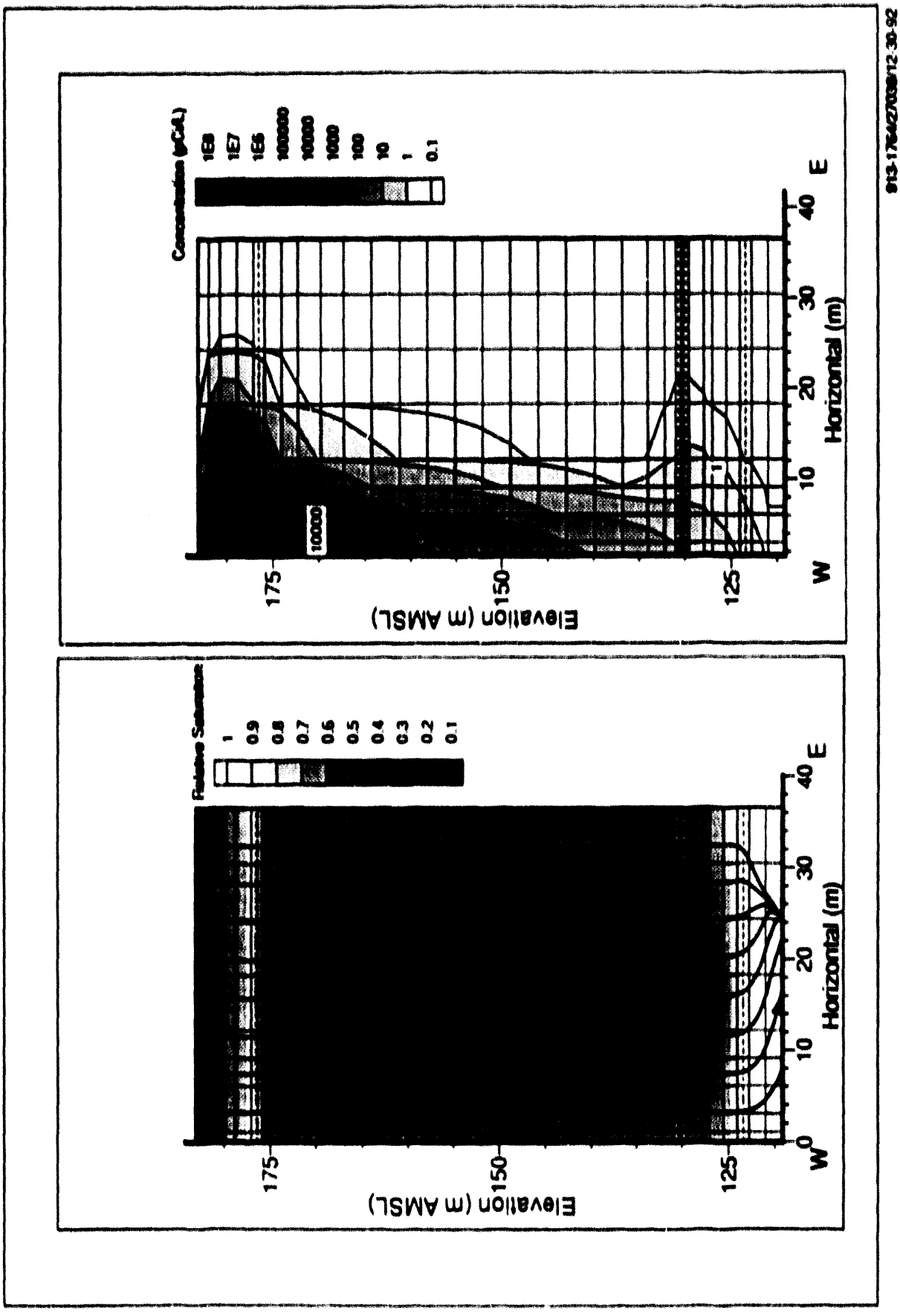
913 1764/44411/1-11-93

Figure 5-47. Concentration of Tc-99 in the Unconfined Aquifer With Time for Scenario B.



913-176427091/12-30-92

Figure 5-48. Relative Saturation, Streamlines, and Total U Concentration for Scenario B at Cessation of Discharge.



913-176402/030/12-30-92

Figure 5-49. Relative Saturation, Streamlines, and Total U Concentration for Scenario B after 37 yr.

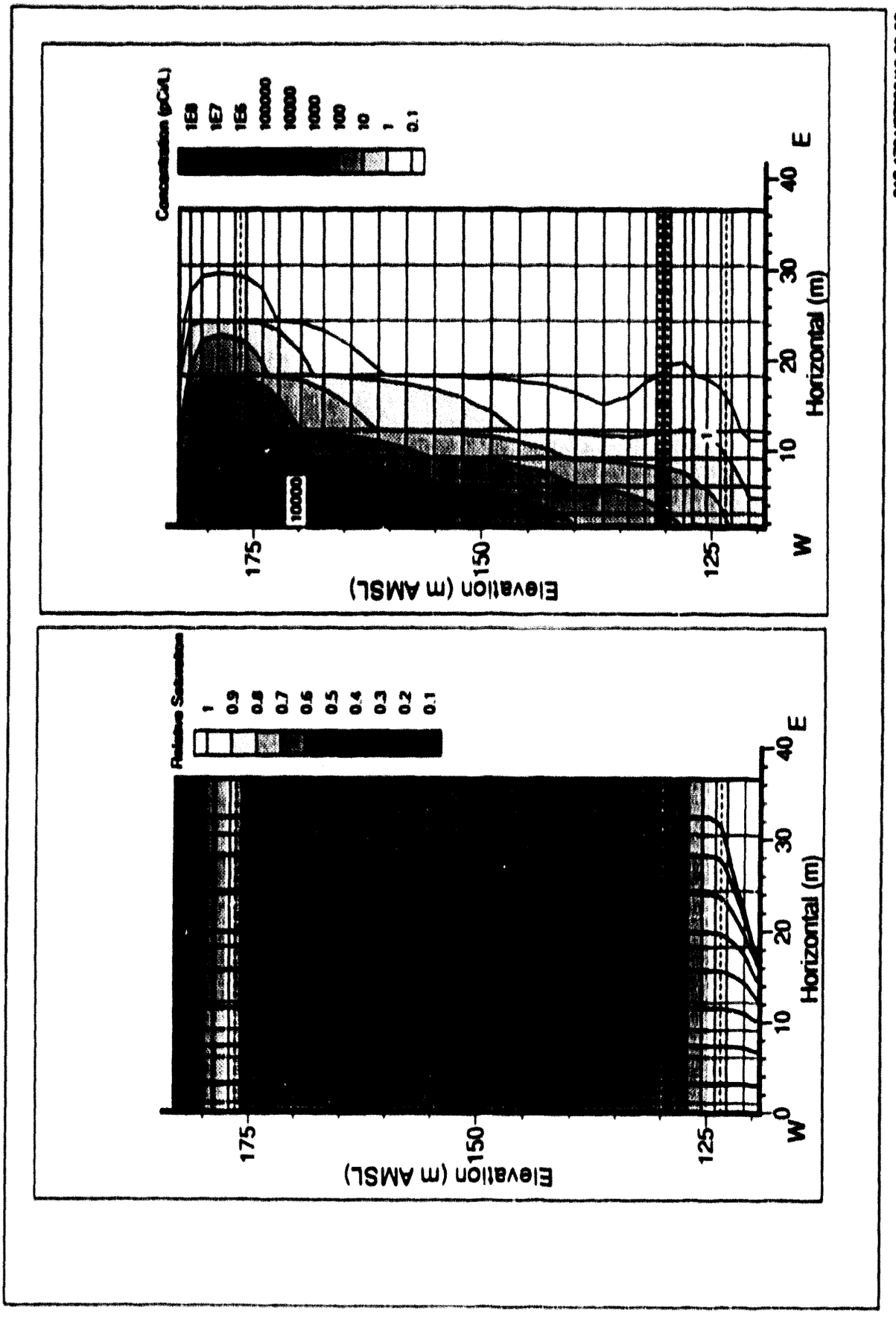


Figure 5-30. Relative Saturation, Streamlines, and Total U Concentration for Scenario B after 150 yr.

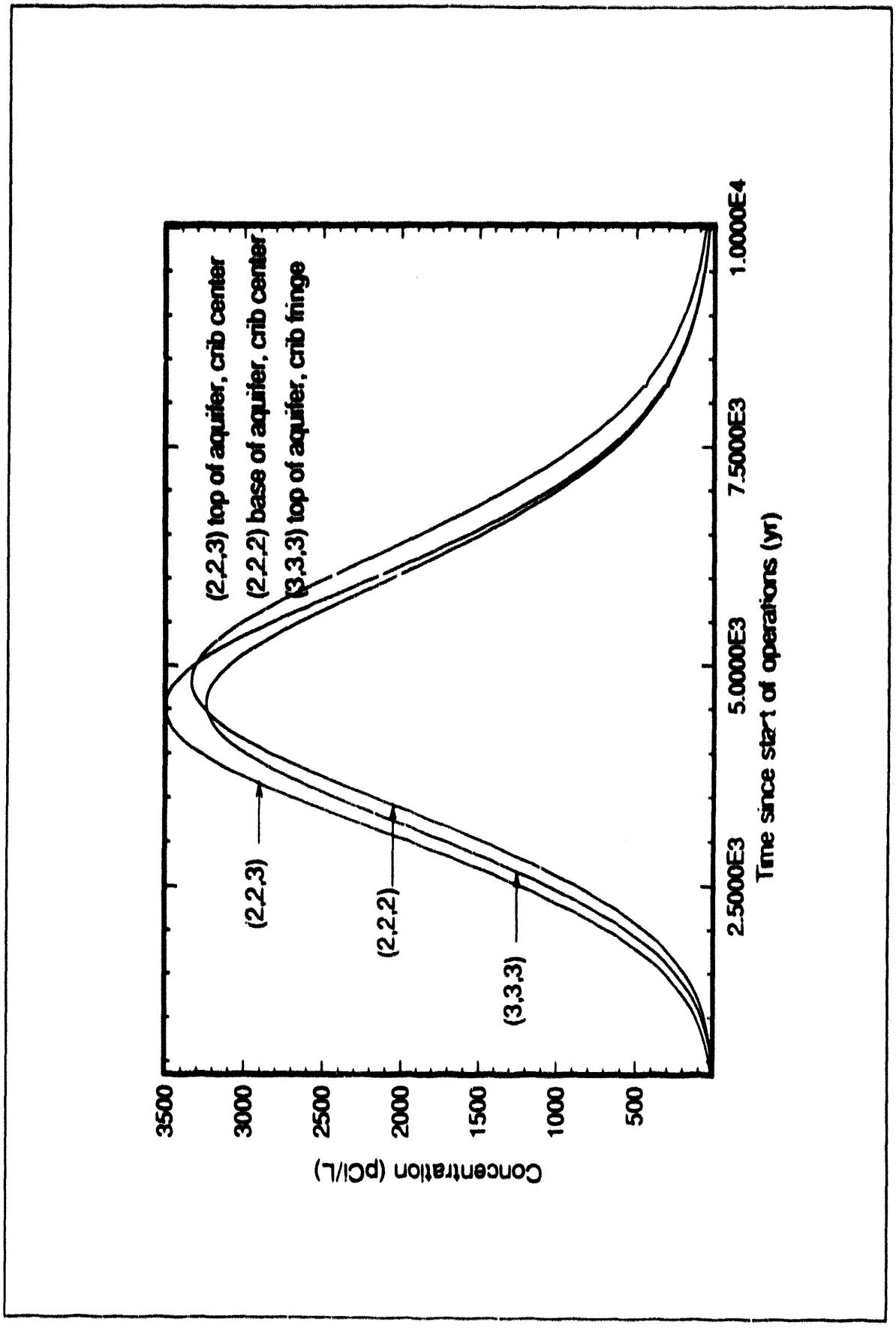
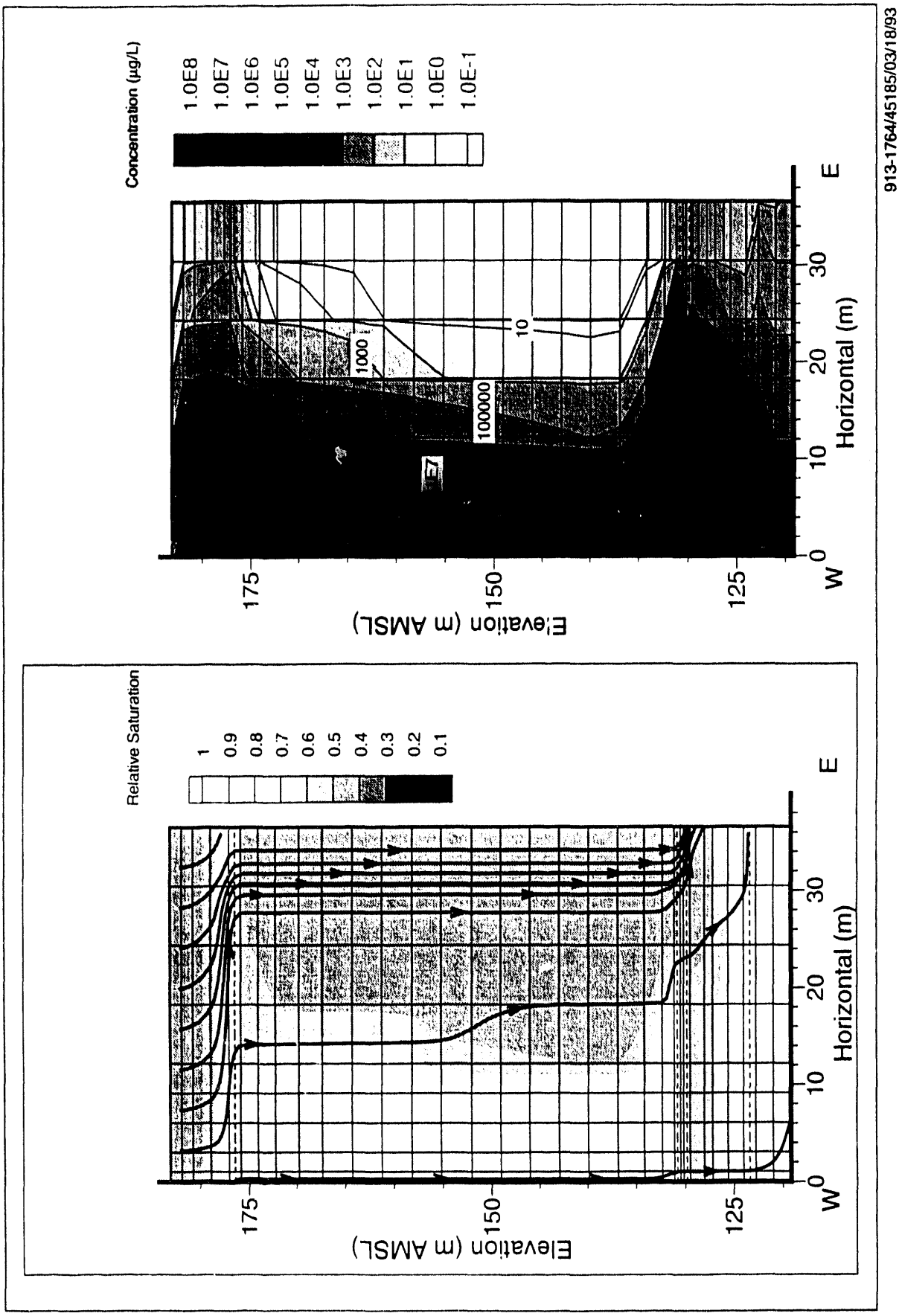
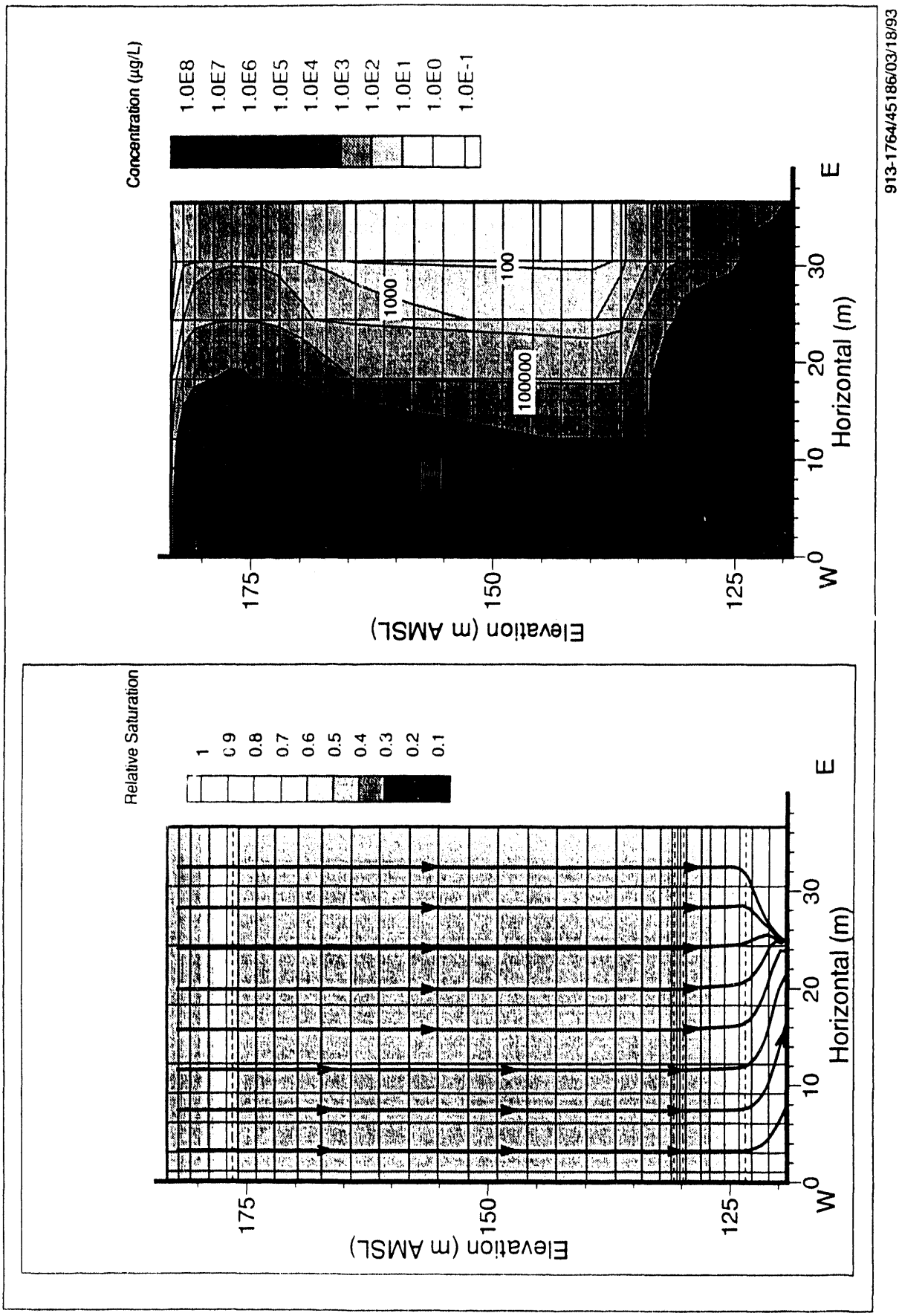


Figure 5-51. Concentration of Total U in the Unconfined Aquifer With Time for Scenario B.



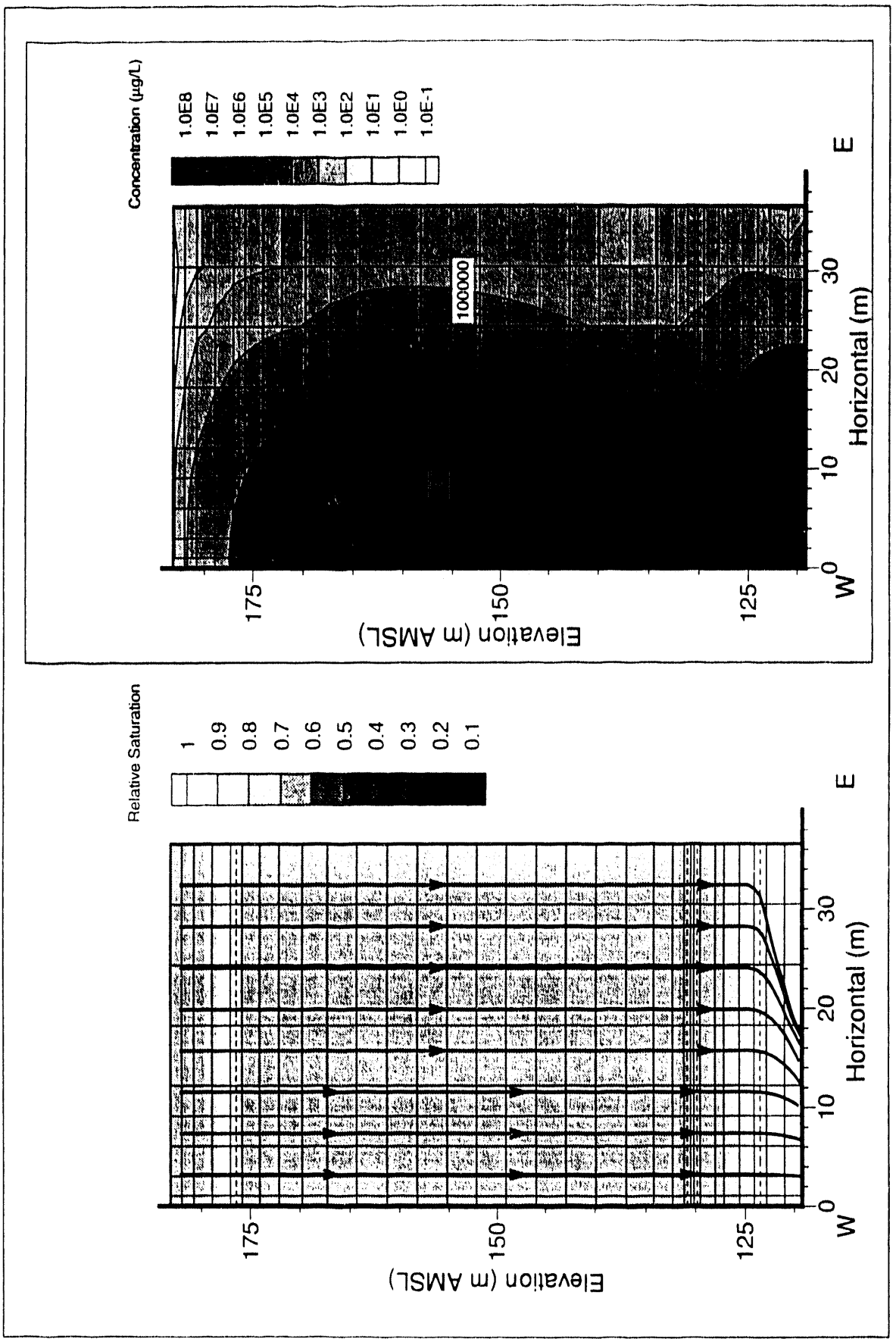
913-1764/45185/03/18/93

Figure 5-52. Relative Saturation, Streamlines, and Nitrate Concentration for Scenario B at Cessation of Discharge.



913-17764/45186/03/18/93

Figure 5-53. Relative Saturation, Streamlines, and Nitrate Concentration for Scenario B after 37 yr.



913-1764/45187/03/18/93

Figure 5-54. Relative Saturation, Streamlines, and Nitrate Concentration for Scenario B after 300 yr.

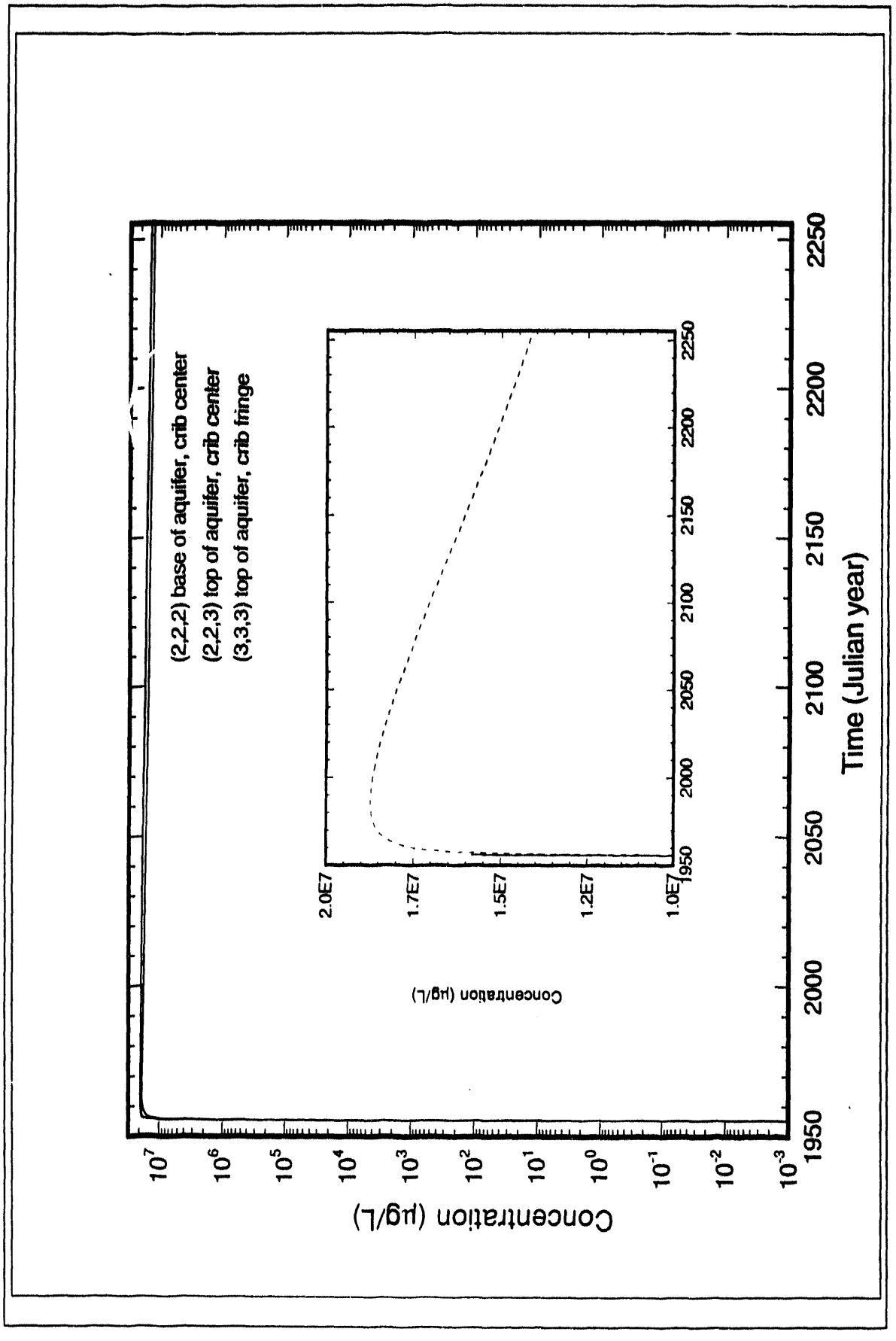


Figure 5-55. Concentration of Nitrate in the Unconfined Aquifer with time for Scenario B.

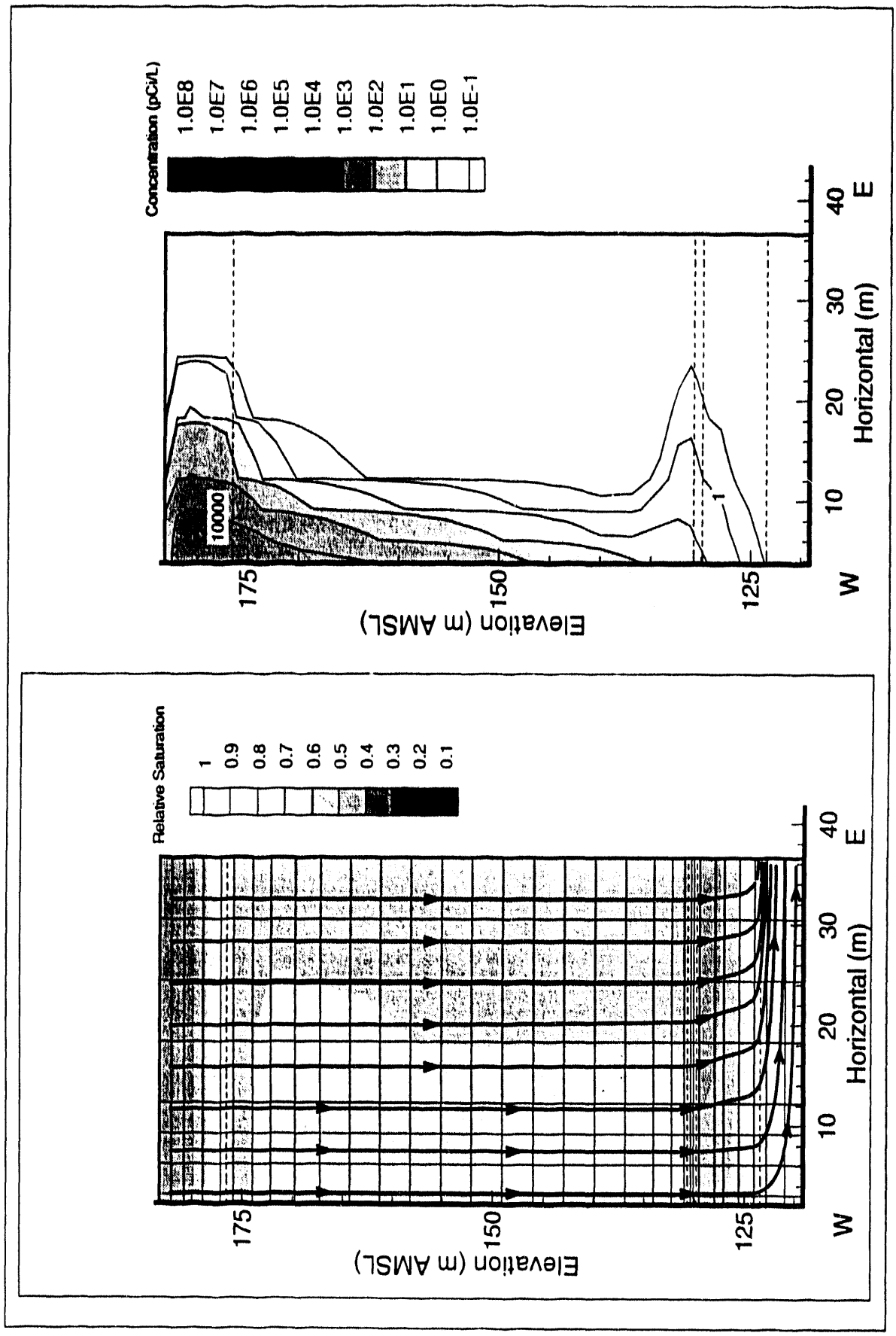


Figure 5-56. Relative Saturation, Streamlines, and Total U Concentration for Scenario C at Cessation of Discharge.

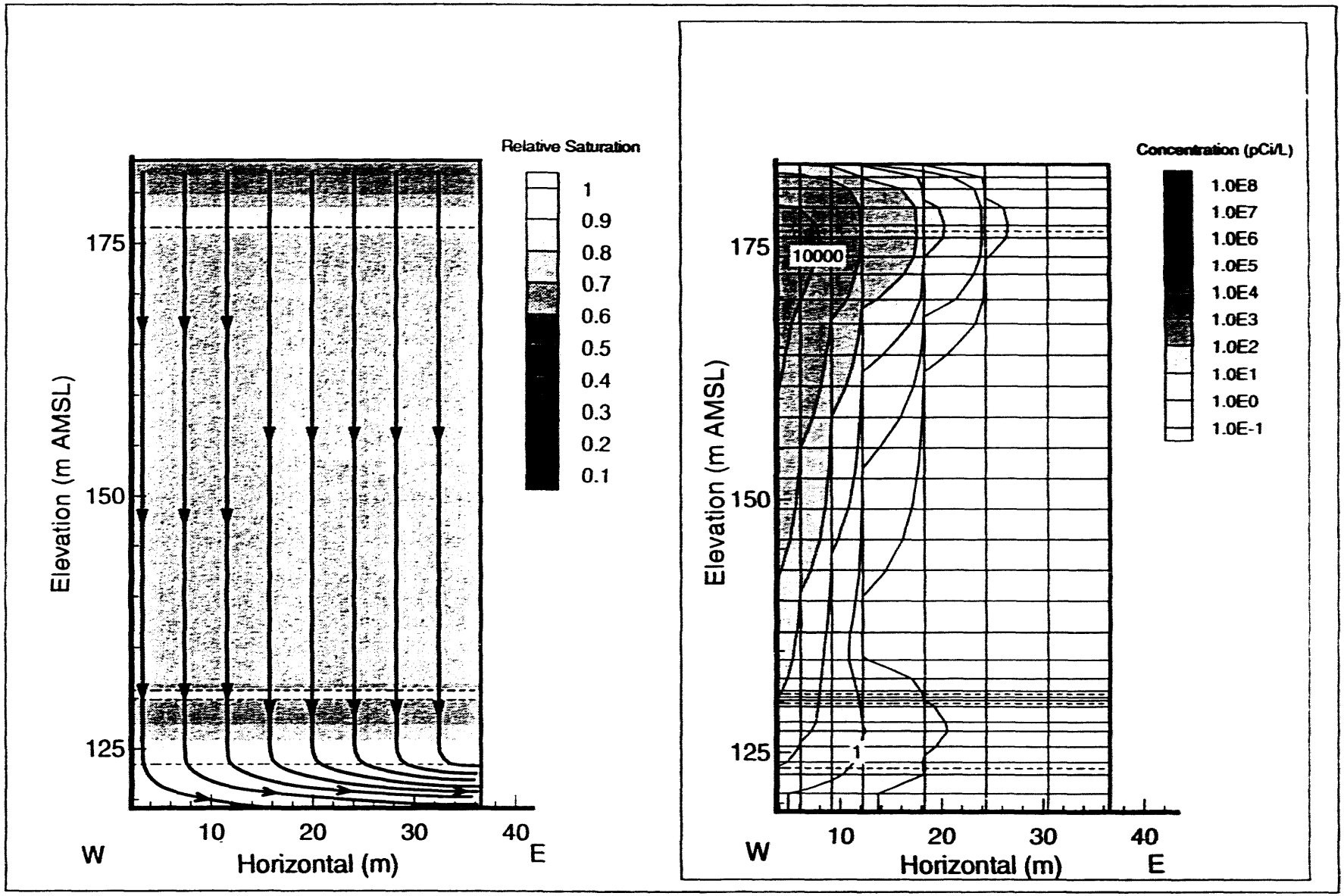
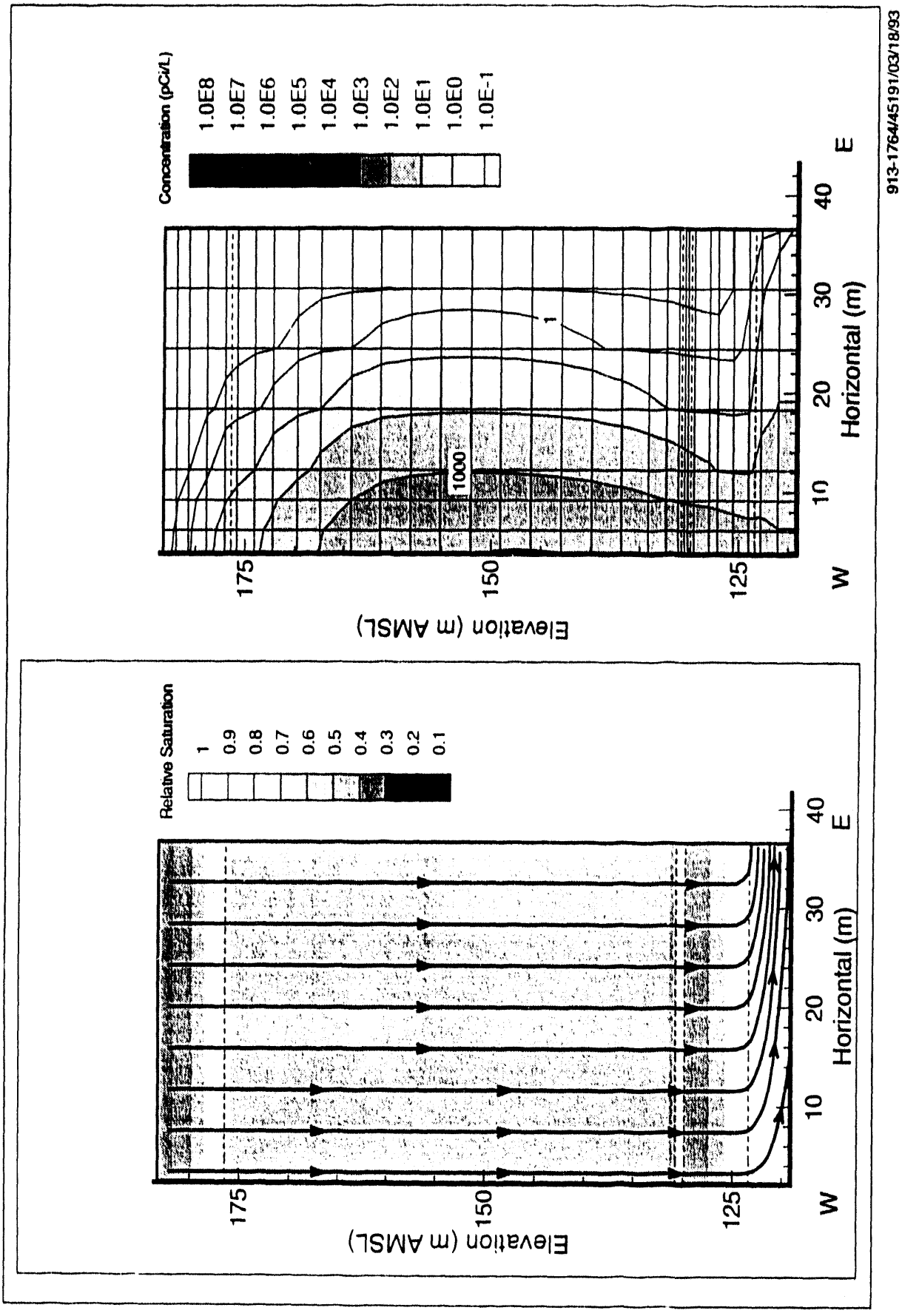


Figure 5-57. Relative Saturation, Streamlines, and Total U Concentration for Scenario C after 37 yr.



913-1764/45191/03/18/93

Figure 5-58. Relative Saturation, Streamlines, and Total U Concentration for Scenario C after 300 yr.

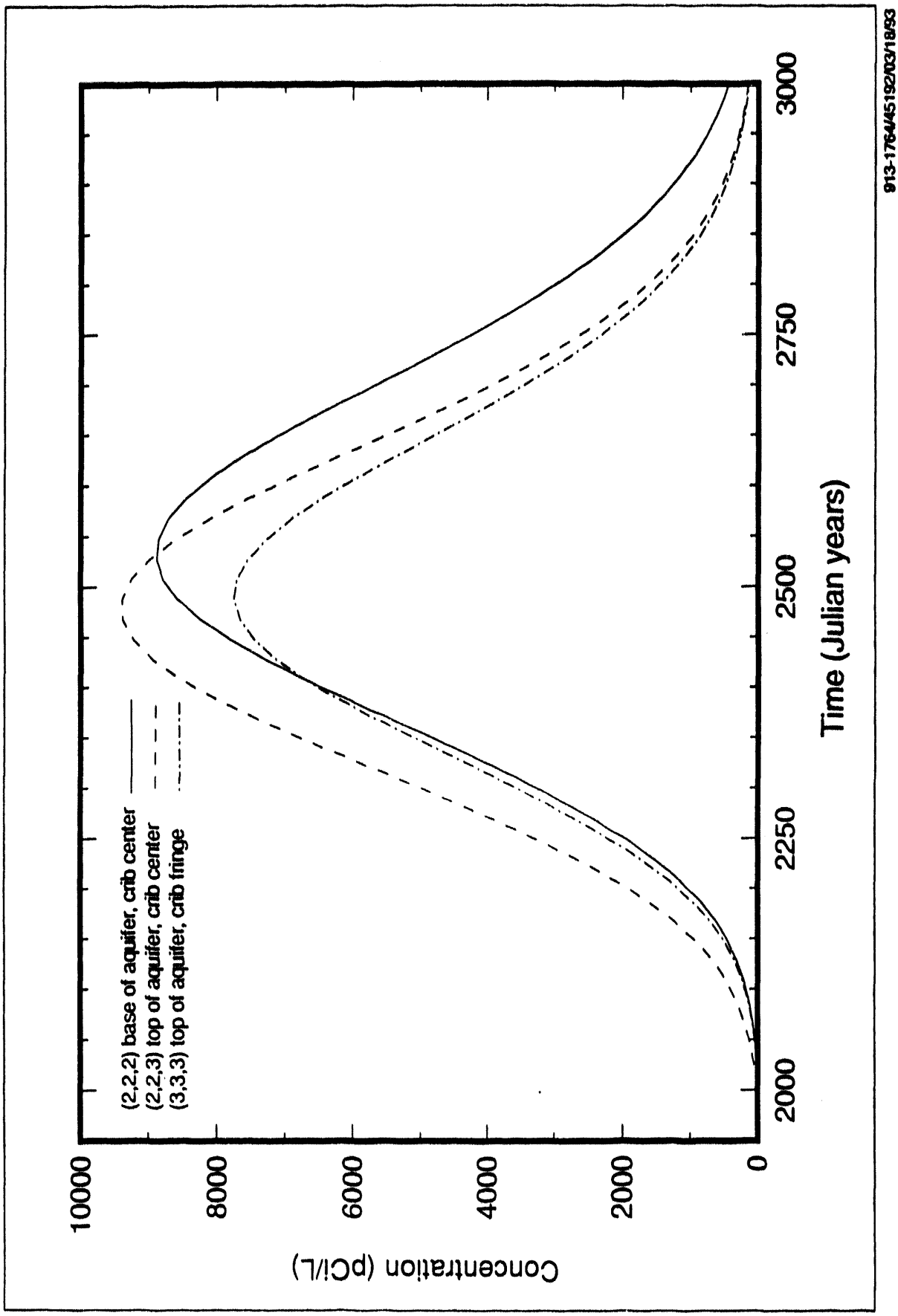


Figure 5-59. Concentration of Total U in the Unconfined Aquifer with time for Scenario C

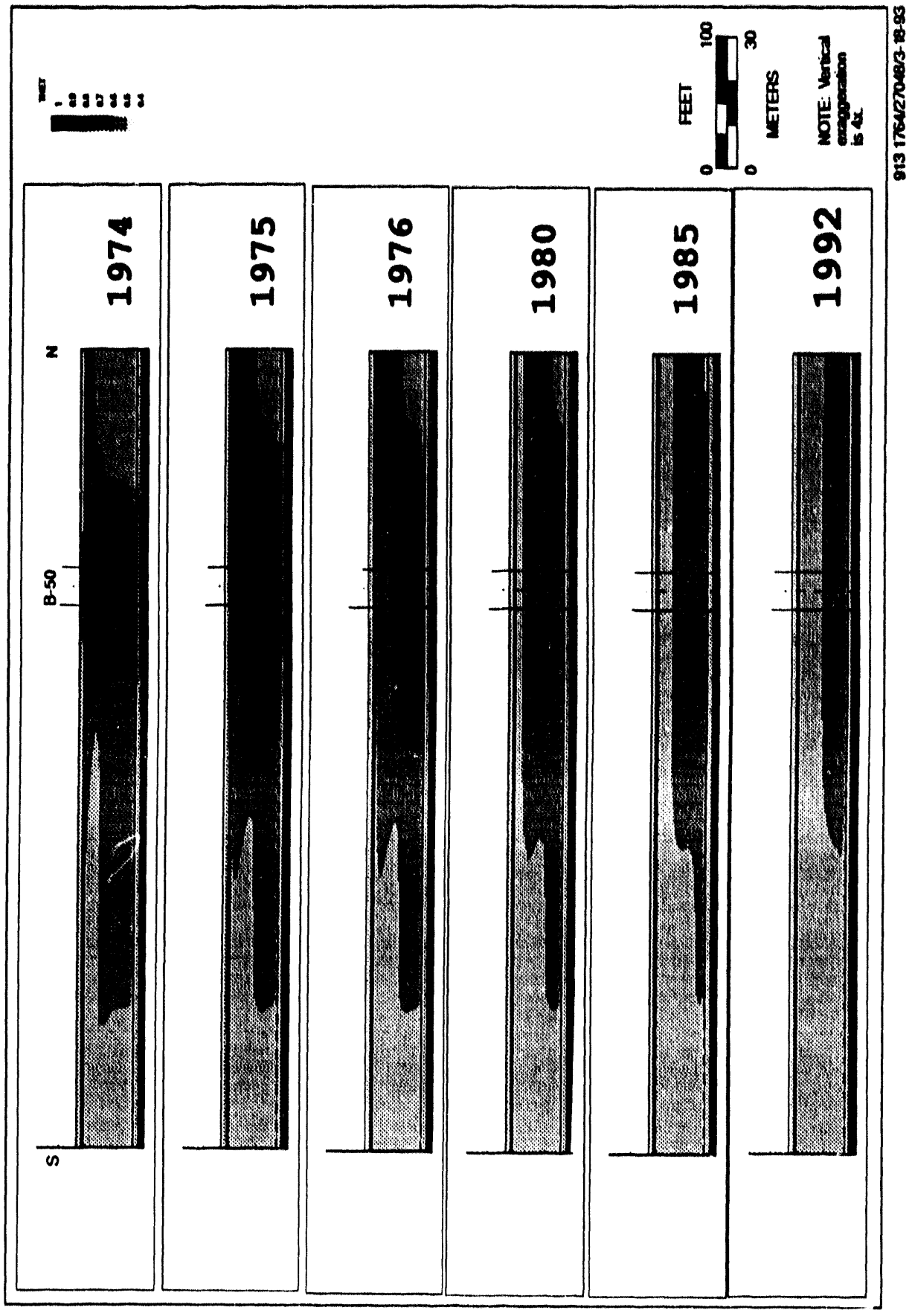


Figure 5-60. Relative Saturation in Cross-Section for Scenario D at 1974, 1975, 1976, 1980, 1985, and 1992.

Table 5-1. Physical Properties of the Organic Contaminants of Potential Concern at the 200-BP-1 Operable Unit.

Contaminant	Mole wt. (g/mole)	Density (g/cm^3)	Melting Point (°C)	Boiling Point (°C)	Vapor Pressure (mm Hg)	Solubility in Water (mg/L)	Henry's k (atm m^3/mole)	Log K_{ow}	K_{oc} (ml/g)	$T^{1/2}$ Soil (d)	$T^{1/2}$ Chem Water (d)	$T^{1/2}$ Air (d)
PCBs - 1232 - 1254	232 328	1.26 1.62	–	310 370	4.1E-03 7.7E-05	– 0.012	– 2.0E-03	5.1 6.5	– 530,000	long	variable	5 to 80
tributyl phosphate	266.3	0.98	-79	292	–	60	–	–	–	–	–	–

k - Henry's law constant - a measure of the extent of chemical partitioning between air and water at equilibrium.
K_{ow} - a measure of the extent of chemical partitioning between water and octanol at equilibrium.
K_{oc} - a measure of the extent of chemcial partitioning between organic carbon and water at equilibrium.
T½ - media specific half-life which is a relative measure of the persistence of a chemical in a given medium.

Sources: SRC 1991; Sax and Lewis 1987; EPA 1988a.

Table 5-2. Sieve Analyses Samples Used in Calculating the Soil Erodibility Index.

Soil Zone	Borehole No.	Well No.	Sample Depth (m)	% Diameter > 0.84 mm
Near-Surface Soils	45A	299-E33-318	0.9	21
	46B	299-E33-299	0.9	46
	49A	299-E33-302	0.9	79
	49B	299-E33-312	2.1	65
	57A	299-E33-304	3.4	65
	61A[a]	299-E33-307	1.7	81
	61A[a]	299-E33-307	2.7	74
			Average =	62
Subsurface Infiltration Gravels/Soils	57A	299-E33-304	4.6	86
	61A	299-E33-307	3.0	92
	61A	299-E33-307	3.7	83
	61A	299-E33-307	4.1	82
			Average =	86
Hanford Site Near Surface Soils	NA	299-E33-38	1.2	56
	NA	299-E33-39	1.2	73
	NA	299-E33-40	1.8	77
	NA	699-50-53B	1.1	86
	NA	699-52-54	1.1	56
	NA	699-52-57	0.6	62
	NA	699-52-57	1.8	76
	NA	699-55-55	0.3	27
	NA	699-55-55	0.6	84
			Average =	66

NA = Not Applicable

[a]Hydrometer analyses results available.

Table 5-3. Values for Variables Used in Calculating Emission Factors.

Source Area	Soil Zone	A[a]	I'[b] ($g/s-m^2$)	K	L'[b]	V'	Q_p ($g/s-m^2$)
216-B-43 - 216-B-50	Near-Surface Soils	0.0575	6.56×10^{-4}	1.0	0.3	1.0	1.063×10^{-6}
	Subsurface Infiltration Gravels/Soils	0.02	3.45×10^{-5}	1.0	0.1	1.0	6.485×10^{-11}
216-B-57	Near-Surface Soils	0.0575	6.56×10^{-4}	1.0	0.2	1.0	7.085×10^{-9}
	Subsurface Infiltration Gravels/Soils	0.02	3.45×10^{-5}	1.0	0.1	1.0	6.485×10^{-11}
Study Area	Near Surface Soils	0.04	8.63×10^{-5}	0.5	1.0	0.5	8.106×10^{-10}

A = Fraction of suspended particulate matter (particle diameters less than 30 µm or 1×10^{-4} ft) (dimensionless).
I' = Soil erodibility factor (mass/area-time).
K = Surface roughness factor (dimensionless).
L' = Unsheltered field width factor (dimensionless).
V' = Vegetative cover factor (dimensionless).

[a]From grain size distribution plots and Baskett 1983.
[b]From Cowherd et al. 1974.

Table 5-4. Decay Factors Used to Calculate Future Radionuclide Upper Confidence Limit Concentrations in the Year 2018.

Radionuclide	Half-life[a] (yr)	Decay Factor[b]
Sb-125	2.8	1.6E-03
Cs-137	3.0E+01	5.5E-01
Co-60	5.3	3.3E-02
Pu-238	8.8E+01	8.1E-01
Pu-239	2.4E+04	1.0
Ra-226	1.6E+03	1.0
Sr-90	2.9E+01	5.4E-01
Tc-99	2.1E+05	1.0
Th-228	1.9	1.0[c]
U-238	4.5E+09	1.0

[a]Health Effects Assessment Summary Tables (HEAST; EPA 1992b).

[b]Decay factors calculated assuming 26 yr between time of sample collection (1992) and beginning of future exposure scenarios (2018).

[c]Although Th-228 has a short half-life, it is assumed to be in equilibrium with its parent, Th-232 (half-life = 1.4E+10 yr).

Table 5-5. Contaminants of Potential Concern Future UCL Soil and Airborne Concentrations (Year 2018). (Page 1 of 3).

Contaminant	Source Area		Source Soil Zone		Source Soil UCL		Airborne UCL Conc.[b]				Figure[c]
	216-B-43 - 216-B-50	216-B-57	Near-Surface	Infil. Gravel[a]	Conc.	Unit	0.0001 $\mu g/m^3$	0.01 $\mu g/m^3$	1.0 $\mu g/m^3$	Unit	
ORGANICS											
PCB	✓			✓	170	µg/kg	1.7E-11	1.7E-09	1.7E-07	$\mu g/m^3$	5-2
		✓		✓	480	µg/kg	4.8E-11	4.8E-09	4.8E-07	$\mu g/m^3$	5-4
TBP	✓			✓	16,000	µg/kg	1.6E-09	1.6E-07	1.6E-05	$\mu g/m^3$	5-2
NONRADIONUCLIDE INORGANICS											
Cadmium	✓			✓	1.0	mg/kg	1.0E-13	1.0E-11	1.0E-09	mg/m^3	5-2
		✓		✓	0.49	mg/kg	4.9E-14	4.9E-12	4.9E-10	mg/m^3	5-4
Nickel	✓			✓	42	mg/kg	4.2E-12	4.2E-10	4.2E-08	mg/m^3	5-2
		✓		✓	9.1	mg/kg	9.1E-13	9.1E-11	9.1E-09	mg/m^3	5-4
RADIONUCLIDES											
Co-60	✓			✓	1.4	pCi/g	1.4E-10	1.4E-08	1.4E-06	pCi/m^3	5-2
Cs-137	✓		✓		13	pCi/g	1.3E-09	1.3E-07	1.3E-05	pCi/m^3	5-1
	✓			✓	1,100,000	pCi/g	1.1E-04	1.1E-02	1.1E-00	pCi/m^3	5-2
		✓	✓		0.27	pCi/g	2.7E-11	2.7E-09	2.7E-07	pCi/m^3	5-3
		✓		✓	19,000	pCi/g	1.9E-06	1.9E-04	1.9E-02	pCi/m^3	5-4
Pu-238	✓			✓	26	pCi/g	2.6E-09	2.6E-07	2.6E-05	pCi/m^3	5-2
		✓		✓	0.0097	pCi/g	9.7E-13	9.7E-11	9.7E-09	pCi/m^3	5-4

Table 5-5. Contaminants of Potential Concern Future UCL Soil and Airborne Concentrations (Year 2018). (Page 2 of 3).

Contaminant	Source Area		Source Soil Zone		Source Soil UCL		Airborne UCL Conc.[b]				Figure[c]
	216-B-43 - 216-B-50	216-B-57	Near-Surface	Infil. Gravel[a]	Conc.	Unit	0.0001 $\mu g/m^3$	0.01 $\mu g/m^3$	1.0 $\mu g/m^3$	Unit	
Pu-239	✓			✓	0.26	pCi/g	2.6E-11	2.6E-09	2.6E-07	pCi/m^3	5-2
		✓		✓	0.0063	pCi/g	6.3E-13	6.3E-11	6.3E-09	pCi/m^3	5-4
Pu-239/240[d]	✓			✓	1,100	pCi/g	1.1E-07	1.1E-05	1.1E-03	pCi/m^3	5-2
Sb-125	✓			✓	0.051	pCi/g	5.1E-12	5.1E-10	5.1E-08	pCi/m^3	5-2
Sr-90	✓		✓		0.97	pCi/g	9.7E-11	9.7E-09	9.7E-07	pCi/m^3	5-1
	✓			✓	1,400,000	pCi/g	1.4E-04	1.4E-02	1.4E+00	pCi/m^3	5-2
		✓	✓		0.12	pCi/g	1.2E-11	1.2E-09	1.2E-07	pCi/m^3	5-3
		✓		✓	23	pCi/g	2.3E-09	2.3E-07	2.3E-05	pCi/m^3	5-4
Ra-226	✓		✓		2.0	pCi/g	2.0E-10	2.0E-08	2.0E-06	pCi/m^3	5-1
	✓			✓	6.9	pCi/g	6.9E-10	6.9E-08	6.9E-06	pCi/m^3	5-2
		✓	✓		0.98	pCi/g	9.8E-11	9.8E-09	9.8E-05	pCi/m^3	5-3
		✓		✓	17	pCi/g	1.7E-09	1.7E-07	1.7E-05	pCi/m^3	5-4
Tc-99	✓			✓	88	pCi/g	8.8E-09	8.8E-07	8.8E-05	pCi/m^3	5-2
		✓		✓	50	pCi/g	5.0E-09	5.0E-07	5.0E-05	pCi/m^3	5-4
Th-228	✓		✓		0.65	pCi/g	6.5E-11	6.5E-09	6.5E-07	pCi/m^3	5-1
	✓			✓	0.85	pCi/g	8.5E-11	8.5E-09	8.5E-07	pCi/m^3	5-2
		✓	✓		0.74	pCi/g	7.4E-11	7.4E-09	7.4E-07	pCi/m^3	5-3
		✓		✓	2.1	pCi/g	2.1E-10	2.1E-08	2.1E-06	pCi/m^3	5-4

Table 5-5. Contaminants of Potential Concern Future UCL Soil and Airborne Concentrations (Year 2018). (Page 3 of 3).

Contaminant	Source Area		Source Soil Zone		Source Soil UCL		Airborne UCL Conc.[b]				Figure[c]
	216-B-43 - 216-B-50	216-B-57	Near-Surface	Infil. Gravel[a]	Conc.	Unit	0.0001 $\mu g/m^3$	0.01 $\mu g/m^3$	1.0 $\mu g/m^3$	Unit	
Total U	✓		✓		0.77	pCi/g	7.7E-11	7.7E-09	7.7E-07	pCi/m^3	5-1
	✓			✓	64	pCi/g	6.4E-09	6.4E-07	6.4E-05	pCi/m^3	5-2
		✓	✓		1.4	pCi/g	1.4E-10	1.4E-08	1.4E-06	pCi/m^3	5-3
		✓		✓	11	pCi/g	1.1E-09	1.1E-07	1.1E-05	pCi/m^3	5-4

UCL = Upper Confidence Limit.

[a]Subsurface infiltration gravel/soil zone.
[b]Airborne contaminant concentrations corresponding to airborne particulate concentrations of 0.0001, 0.01, and 1.0 $\mu g/m^3$.
[c]Figure number depicting airborne particulate concentration contours for a particular combination of source area and soil zone.
[d]A single Pu-239/240 soil concentration value is available for the subsurface infiltration gravel soil zone within the 216-B-57 source area.

Table 5-6. Contaminants of Potential Concern Future UCL Soil and Ground Surface Concentrations (Year 2018). (Page 1 of 3).

Contaminant	Source Area		Source Soil Zone		Source Soil UCL		Ground Surface UCL Conc.[b]				Figure[c]
	216-B-43 - 216-B-50	216-B-57	Near-Surface	Infil. Gravel[a]	Conc.	Unit	10^{-6} Mixing Factor	10^{-4} Mixing Factor	10^{-1} Mixing Factor	Unit Mixing Factor	
ORGANICS											
PCB	✓			✓	170	µg/kg	1.7E-04	1.7E-02	1.7E+01	µg/kg	5-6
		✓		✓	480	µg/kg	4.8E-04	4.8E-02	4.8E+01	µg/kg	5-8
TBP	✓			✓	16,000	µg/kg	1.6E-02	1.6E+00	1.6E+03	µg/kg	5-6
NONRADIONUCLIDE INORGANICS											
Cadmium	✓			✓	1.0	mg/kg	1.0E-06	1.0E-04	1.0E-01	mg/kg	5-6
		✓		✓	0.49	mg/kg	4.9E-07	4.9E-05	4.9E-02	mg/kg	5-8
Nickel	✓			✓	42	mg/kg	4.2E-05	4.2E-03	4.2E+00	mg/kg	5-6
		✓		✓	9.1	mg/kg	9.1E-06	9.1E-04	9.1E-01	mg/kg	5-8
RADIONUCLIDES											
Co-60	✓			✓	1.4	pCi/g	1.4E-06	1.4E-04	1.4E-01	pCi/g	5-6
Cs-137	✓		✓		13	pCi/g	1.3E-05	1.3E-03	1.3E+00	pCi/g	5-5
	✓			✓	1,100,000	pCi/g	1.1E+00	1.1E+02	1.1E+05	pCi/g	5-6
		✓	✓		0.27	pCi/g	2.7E-07	2.7E-05	2.7E-02	pCi/g	5-7
		✓		✓	19,000	pCi/g	1.9E-02	1.9E+00	1.9E+03	pCi/g	5-8
Pu-238	✓			✓	26	pCi/g	2.6E-05	2.6E-03	2.6E+00	pCi/g	5-6
		✓		✓	0.0097	pCi/g	9.7E-09	9.7E-07	9.7E-04	pCi/g	5-8

Table 5-6. Contaminants of Potential Concern Future UCL Soil and Ground Surface Concentrations (Year 2018). (Page 2 of 3).

Contaminant	Source Area		Source Soil Zone		Source Soil UCL		Ground Surface UCL Conc.[b]				Figure[c]
	216-B-43 - 216-B-50	216-B-57	Near-Surface	Infil. Gravel[a]	Conc.	Unit	10^{-6} Mixing Factor	10^{-4} Mixing Factor	10^{-1} Mixing Factor	Unit Mixing Factor	
Pu-239	✓			✓	0.26	pCi/g	2.6E-07	2.6E-05	2.6E-02	pCi/g	5-6
		✓		✓	0.0063	pCi/g	6.3E-09	6.3E-07	6.3E-04	pCi/g	5-8
Pu-239/240[d]	✓			✓	1100	pCi/g	1.1E-03	1.1E-01	1.1E+02	pCi/g	5-6
Sb-125	✓			✓	0.051	pCi/g	5.1E-08	5.1E-06	5.1E-03	pCi/g	5-6
Sr-90	✓		✓		0.97	pCi/g	9.7E-07	9.7E-05	9.7E-02	pCi/g	5-5
	✓			✓	1,400,000	pCi/g	1.4E+00	1.4E+02	1.4E+05	pCi/g	5-6
		✓	✓		0.12	pCi/g	1.2E-07	1.2E-05	1.2E-02	pCi/g	5-7
		✓		✓	23	pCi/g	2.3E-05	2.3E-03	2.3E+00	pCi/g	5-8
Ra-226	✓		✓		2.0	pCi/g	2.0E-06	2.0E-04	2.0E-01	pCi/g	5-5
	✓			✓	6.9	pCi/g	6.9E-06	6.9E-04	6.9E-01	pCi/g	5-6
		✓	✓		0.98	pCi/g	9.8E-07	9.8E-05	9.8E-02	pCi/g	5-7
		✓		✓	17	pCi/g	1.7E-05	1.7E-03	1.7E+00	pCi/g	5-8
Tc-99	✓			✓	88	pCi/g	8.8E-05	8.8E-03	8.8E+00	pCi/g	5-6
		✓		✓	50	pCi/g	5.0E-05	5.0E-03	5.0E+00	pCi/g	5-8
Th-228	✓		✓		0.65	pCi/g	6.5E-07	6.5E-05	6.5E-02	pCi/g	5-5
	✓			✓	0.85	pCi/g	8.5E-07	8.5E-05	8.5E-02	pCi/g	5-6
		✓	✓		0.74	pCi/g	7.4E-07	7.4E-05	7.4E-02	pCi/g	5-7
		✓		✓	2.1	pCi/g	2.1E-06	2.1E-04	2.1E-01	pCi/g	5-8

Table 5-6. Contaminants of Potential Concern Future UCL Soil and Ground Surface Concentrations (Year 2018). (Page 3 of 3).

Contaminant	Source Area		Source Soil Zone		Source Soil UCL		Ground Surface UCL Conc.[b]				Figure[c]
	216-B-43 - 216-B-50	216-B-57	Near-Surface	Infil. Gravel[a]	Conc.	Unit	10^{-6} Mixing Factor	10^{-4} Mixing Factor	10^{-1} Mixing Factor	Unit Mixing Factor	
Total U	✓		✓		0.77	pCi/g	7.7E-07	7.7E-05	7.7E-02	pCi/g	5-5
	✓			✓	64	pCi/g	6.4E-05	6.4E-03	6.4E+00	pCi/g	5-6
		✓	✓		1.4	pCi/g	1.4E-06	1.4E-04	1.4E-01	pCi/g	5-7
		✓		✓	11	pCi/g	1.1E-05	1.1E-03	1.1E+00	pCi/g	5-8

UCL = Upper Confidence Limit.

[a]Subsurface infiltration gravel/soil zone.
[b]Ground surface UCL concentrations are determined from the ratio of 200-BP-1 source dust fallout rate to the entire study area dust fallout rate times the 200-BP-1 source UCL concentrations.
[c]Figure number depicting airborne particulate concentration contours for a particular combination of source area and soil zone.
[d]A single Pu-239/240 soil concentration value is available for the subsurface infiltration gravel soil zone within the 216-B-57 source area.

Table 5-7. Liquid Effluent Discharge Rates and Periods of Discharge for Scenarios A, B and C.

Crib	Discharge Period		Discharge Volume (L)	Infiltration Rate (m/yr)
	Begin	End		
216-B-43	11/54	11/54	25,440,000	304
216-B-44	11/54	3/55	13,440,000	161
216-B-45	4/55	6/55	19,680,000	235
216-B-46	9/55	12/55	20,160,000	240
216-B-47	9/55	9/55	44,520,000	532
216-B-48	11/55	11/55	49,080,000	587
216-B-49	11/55	12/55	40,200,000	481
216-B-50	1/65	1/74	6,120,000	73

Table 5-8. Source Inventory and Effluent Concentration of Contaminants Used for Vadose Zone Modeling.

Contaminant	Source Inventory[a]	Effluent Concentration[b]
TBP	Unknown	Unity
Co-60	21.14 Ci	1.074×10^{6} pCi/L
Cs-137	5,916 Ci	3.007×10^{8} pCi/L
Pu-238, 239, 239/240	5.8 Ci	2.947×10^{5} pCi/L
Sr-90	11,150 Ci	5.66×10^{8} pCi/L
Tc-99	Unknown	Unity
Total U	0.424 Ci	2.153×10^{4} pCi/L
Nitrate	6×10^{6} kg	3.049×10^{5} mg/L

[a]Maximum inventory of any BY crib from Appendix H.
[b]Liquid amount assumed equal to that of crib 216-B-45 (19.7 million liter).

Table 5-9. Soil Properties for the Four Layers.

Soil Layer	Depth[a] (m)		van Genuchten		K_S (cm/s)	Porosity			S_S (m^{-1})	ρ_D (kg/m^3)
	Top	Bottom	α	n		Effective	Total	Diffusive		
Sandy Gravel	0	1.8	1.87	1.57	2.80×10^{-3}	0.266	0.321	0.321	0.3	1980
Sand	1.8	15.8	0.011	1.23	5.00×10^{-4}	0.274	0.274	0.274	0.3	1810
Silt	15.8	16.2	5.41	1.66	2.09×10^{-6}	0.392	0.392	0.392	0.4	1610
Silty Sandy Gravel	16.2	19.2	1.03	1.36	4.20×10^{-3}	0.288	0.288	0.288	0.3	2010

K_S = Saturated hydraulic conductivity. Isotropic for Scenarios A and B. Vertical for Scenario C, with horizontal equal to fives times vertical.

S_S = Specific storativity.

ρ_D = Dry bulk density.

[a]Depth measured from bottom of crib.

Table 5-10. Contaminant Distribution Coefficients and Half-Lives.

Contaminant	Half-Life		K_d (mg/L)			
	yr	Reference	Used	Min.	Max.	Reference
TBP	N/A	N/A	4	None	None	Ames and Serne (1991)
Co-60	5.3	Kocher (1981)	1.94[a]	1.94	200	Strenge and Peterson (1989)
				100	10000	Cantrell and Serne (1992)
Cs-137	30.2	DHEW (1970)	500	500	10000	Cantrell and Serne (1992)
Pu-238,239,239/240	2.41×10^4	DHEW (1970)	100	100	2000	Cantrell and Serne (1992)
Sr-90	28.6	Kocher (1981)	20	5	100	Cantrell and Serne (1992)
Tc-99	2.1×10^5	DHEW (1970)	0.1	0	0.1	None[b]
Total U	4.51×10^9	DHEW (1970)	1	0	3	Cantrell and Serne (1992)
Nitrate	N/A	N/A	0	-	-	-

K_d = Distribution coefficient.
N/A = Not Applicable, assumed to be persistent.

[a]Modeling results indicate that a distribution coefficient of 1.94 mg/L is too large. Additional modeling was performed with an assumed distribution coefficient of 0.5 mg/L.
[b]Preliminary laboratory results from Tc-99 adsorption measurements in 200-BP-1 operable unit soil samples indicate a K_d from 0 to 0.05 mg/L. A K_d value of 0.1 mg/L was chosen for conservatism.

Table 5-11. Results of Comparison of Present-Day Predicted Versus Measured Concentrations.

Contaminant	K_d (ml/g)	Model-Predicted Concentration[a]				Measured Crib Soils		Column Leach Test Maximum Measured Leachate Concentrations	
		C_w	Units	C_s	Units	$C_{s,ucl}$	Unit	$C_{w,l}$	Units
Nitrate	0	$5x10^4$	mg/L	N/A		$1.2x10^3$	mg/kg	$2.0x10^3$	mg/L
Co-60	0.5	$5x10^3$	pCi/L	2.5	pCi/g	42	pCi/g	$2.7x10^3$	pCi/L
Cs-137	500	$1x10^8$	pCi/L	$5x10^6$	pCi/g	$2.0x10^8$	pCi/g	$8.4x10^6$	pCi/L
Sr-90	20	$1x10^8$	pCi/L	$2x10^6$	pCi/g	$2.6x10^8$	pCi/g	$5.2x10^7$	pCi/L
Total U	1	$5x10^4$	pCi/L	50	pCi/g	64	pCi/g	2.210[4]	pCi/L

C_w = Concentration in model predicted water.
C_s = Concentration in model predicted soil = $C_w \times K_d$
$C_{s,ucl}$ = UCL soil concentration presented in Table 4-20 for the crib grouping 216-B-43 through 216-B-50 and the subsurface infiltration gravel/soil zone.
$C_{w,l}$ = Concentration of leachate from column leach test (Task 10) from Gillespie 1992.
N/A = Not applicable with a K_d of 0.
[a]Modeled results are for Scenario B conditions, except for Cs-137 and Sr-90 which are for Scenario A conditions.

6.0 BASELINE RISK ASSESSMENT

The baseline risk assessment is an analysis of the potential adverse health and environmental effects (current and future) which may be caused by hazardous substances at a site under an assumption of no remedial action. This chapter provides an assessment of the threats posed to human health and the environment by the contaminants of potential concern that have been detected at the 200-BP-1 operable unit.

6.1 IDENTIFICATION OF CONTAMINANTS OF POTENTIAL CONCERN

Contaminants of potential concern are identified and discussed in Sections 4.3.1.3 and 4.4. A list of these contaminants is presented in Table 4-20 for both near surface soils (i.e., 0-15 ft) and infiltration gravels/soils (i.e., > 15 ft). Included in this table are the 95% upper confidence limits (UCLs) of the mean soil concentrations for each contaminant at each crib grouping. The UCLs used in this chapter represent the upper confidence limit of the mean concentrations. The use of 95% UCLs is in accordance with RAGS (EPA 1989a) and HSBRAM (DOE-RL 1993b), and provides a conservative estimate of the contaminant concentrations a receptor may encounter. These concentrations are utilized in Sections 6.2 and 6.3 to calculate contaminant intakes and quantify the health risks associated with those intakes.

There are a few instances in which a 95% UCL exceeds an associated maximum detected value. Although RAGS (EPA 1989a) recommends using the maximum detected value for such cases, this risk assessment uses the 95% UCLs to calculate intakes and risks. This is because the sample sizes for these data sets are generally too low to reasonably consider the maximum detected value to be representative of contaminant concentrations, and use of the 95% UCL adds a level of conservatism to compensate for this condition.

6.2 HUMAN HEALTH EVALUATION

6.2.1 Human Exposure Assessment

The purpose of the human exposure assessment is to estimate the magnitude, frequency, duration, and route of exposure to the contaminants of potential concern that human receptor populations may experience. This exposure information can then be integrated with appropriate toxicity information to provide an assessment of the nature and extent of any health threats from the contaminants of potential concern. The exposure assessment considers measured and estimated contaminant concentrations in various environmental media, identifies potential human receptor populations, and determines probable routes of contaminant intake.

6.2.1.1 Identification of Exposure Scenarios. Four exposure scenarios (industrial, residential, recreational, and agricultural) are evaluated in this risk assessment (Table 6-1). The matrix presented in Table 6-1 is based on meeting-minutes of Tri-Party Agreement unit managers. These scenarios are evaluated under both current and assumed future site conditions, and for the following locations: on the 200-BP-1 operable unit, on the Hanford site (but outside of the 200 Area), and off the Hanford site. It should be noted that, according to the TPA, the year 2018 is the earliest time that the Federal Government could release portions of the

Hanford site for non-industrial uses. Hence, the future scenario in this risk assessment is assumed to begin in 2018.

Evaluation of the current scenario recognizes that a clean soil cover currently exists at the 200-BP-1 operable unit (Hayward 1992), and that access to the unit is currently limited. Existence and maintenance of the clean soil cover does not represent a true baseline condition, but is a "limited action" scenario in accordance with recent DOE guidance (DOE 1992). Evaluation of the future scenarios is divided into three possibilities:

- The clean soil cover is maintained and undisturbed
- The near surface soils are exposed and/or excavated
- The infiltration gravels/soils are exposed and deposited on the ground surface.

Considering the depth to the infiltration gravels/soils [approximately 5 m (15 ft)], the last of these possibilities is extremely unlikely. However, it is evaluated in order to provide decision makers with an estimate of the potential human health impacts if such an event is allowed to occur.

The industrial scenario is the only exposure scenario considered viable onsite (within the 200-BP-1 operable unit) for both current and future conditions because of the probability that this area will become a permanent waste management zone (Table 6-1), and is in accordance with the Hanford Future Site Uses Working Group (1992) report.

The industrial scenario is the only operative exposure scenario under current conditions on the Hanford Site, outside of the 200 Areas (i.e., 600 Area lands). However, for the future conditions, the viable exposure scenarios include industrial, residential, recreational, and agricultural.

In the case of the exposure scenarios off of the Hanford Site, all the identified scenarios (i.e., industrial, residential, recreational, and agricultural) are considered to be viable for both current and future conditions (Table 6-1).

6.2.1.2 Identification of Exposure Pathways. The HSBRAM (DOE-RL 1993b) identifies the exposure pathways that are evaluated for the industrial, residential, recreational, and agricultural receptors. However, because the 200-BP-1 operable unit is focused only on soils, exposure pathways that may be associated with potential exposure to either groundwater or surface water at the site are not evaluated in this baseline risk assessment. In addition, the potential leachability of contaminants from subsoils into onsite groundwater and potential migration into the Columbia River will also not be evaluated. This topic will be addressed in a subsequent risk assessment that will focus on the groundwater medium in the 200 East Aggregate Area of the site.

6.2.1.2.1 Current Exposure . Under current conditions, the only viable exposure pathway is external exposure to radionuclides by the industrial workers within the 200-BP-1 operable unit. The clean soil cover at the 200-BP-1 operable unit serves to prevent the workers from coming in direct contact with contaminated soils, eliminating potential ingestion and dermal exposure to soils. The clean soil cover also prevents the potential erosion of contamination soils, thereby eliminating the inhalation exposure pathway under the current scenario. The only potential exposure to current workers on the operable unit is from external radiation that penetrates through the clean soil cover. The workers on the 200-BP-1

operable unit are, however, subjected to naturally occurring and general anthropogenic soil and air constituents which originate from areas both within and outside the Hanford Site. Consequently, available environmental monitoring data are used to evaluate the potential risks to onsite workers from exposure to these contaminants. It is also noteworthy that the potential current risks that may be associated with the workers on the site (but outside the 200-BP-1 operable unit) would be comparatively lower.

6.2.1.2.2 Future Exposure. One of the three possible future settings being evaluated is that the current clean soil cover is maintained and undisturbed. Under this condition, the only operative pathway is external exposure of industrial worker receptors to radiation penetrating the cover on the operable unit.

The two remaining future settings involve the exposure of near surface soils and infiltration gravels/soils. Table 6-2 provides a matrix of the exposure pathways evaluated for each of the four exposure scenarios with contaminated soils exposed.

Industrial Scenario (on operable unit). The soil ingestion, dermal, and external exposure pathways are evaluated only for the future industrial scenario for the 200-BP-1 operable unit because operable unit workers are the only receptor population with the potential for direct access to contaminated soils. Fugitive dust inhalation is also evaluated for the operable unit location.

Industrial (*off* operable unit), Residential, and Agricultural Scenarios. Inhalation of contaminants via fugitive dust is the major pathway evaluated for locations away from the operable unit because receptors at these locations will not have direct access to contaminated soils. The non-inhalation exposure pathways (e.g., soil ingestion, external exposure to radionuclides, and biota ingestion) could arise from the deposition of fugitive dust outside of the 200-BP-1 operable unit and 200 Areas. However, such pathways are not usually evaluated in most risk assessments for the following reasons: (1) dilution of contaminant concentrations being transported via fugitive dust is expected to be so large that the resulting off-site soil concentrations will be extremely small; (2) the industrial scenario on the operable unit is likely to be associated with the greatest risk because receptors on the unit will be exposed to the highest contaminant concentrations; and (3) for many contaminants (e.g., alpha-emitting radionuclides) the inhalation pathway will provide the highest risk. While deposition of fugitive dust is not an issue for most risk assessments, it may be important at the 200-BP-1 operable unit because contaminant concentrations (in the infiltration gravels/soils) are so high that the assumption of insignificant off-site concentrations may not be valid. In addition, the biota ingestion pathway is particularly important for some contaminants (e.g., strontium-90) due to relatively high bioconcentration factors.

Rather than evaluate the human health impacts of fugitive dust deposition for all contaminants of potential concern, such an analysis will be made only for those contaminants for which exposures following dust deposition are particularly important. Contaminants are evaluated if their risk estimates from exposure on the operable unit exceed 10^{-4}, and if the risks associated with exposure to dust deposition are expected to be greater than inhalation risk estimates (which are already evaluated for off-site locations). Additional information on the evaluation of fugitive dust deposition is provided in Section 6.2.3.3.

Recreational Scenario. As with the residential and agricultural scenarios, inhalation of contaminants via fugitive dust is the major pathway evaluated for recreational receptors because these receptors will not have direct access to contaminated soils.

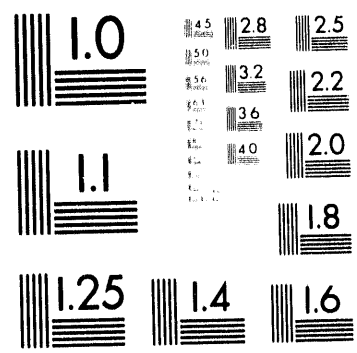
Another possible way for recreational receptors on the Hanford Site to come in contact with operable unit contaminants is through the potential ingestion of contaminated game birds and/or animals. However, mule deer are not common within the 200 Area because of the high barbed-wire-topped fence surrounding the area. In addition, a conservative mule deer home range is approximately 510 ha (1,260 ac) (Paustenbach 1989), but the 200-BP-1 operable unit encompasses an area of approximately 10 ha (25 ac). Assuming that the deer's browsing rate is uniform throughout the range, a mule deer foraging in the vicinity of the operable unit would obtain only 2% of its total diet from the operable unit. Furthermore, the probability that a hunter consumes a significant number of game birds that forage on a permanent waste management zone will be exceedingly small. Therefore, the game ingestion pathway is assumed to be minor, relative to other potential exposure pathways at this site, and is not considered in this risk assessment.

6.2.1.3 Quantification of Human Exposures. The exposure assessment provides quantitative exposure factors for the pathways that have been identified for each receptor population. An exposure point concentration (i.e., a contaminant concentration to which a receptor is subjected over the exposure period) is estimated and used with exposure parameters (e.g., contact rate, body weight, and exposure frequency) to determine an intake. The following sections describe the assumptions and calculations used to quantify exposure intakes for the industrial, residential, recreational and agricultural receptor populations. Intake values are provided for a contaminant only if there is a toxicity value for a specific exposure route which permits a quantitative evaluation of that contaminant. For example, fugitive dust intake values for PCBs and tributyl phosphate are not calculated because these compounds are either not considered to be toxic via the inhalation route, or toxicity values have not been determined.

6.2.1.3.1 Exposure Point Concentrations. The 95% UCL soil concentrations (Table 4-20) are adopted as the exposure point concentrations used to evaluate soil ingestion, dermal exposure, and external exposure to contaminated soils. As described in Section 4.4, separate 95% UCLs were calculated for near surface soils and infiltration gravels/soils, and for each crib grouping.

As mentioned in Section 6.2.1, a clean soil cover is currently in place at the 200-BP-1 operable unit; hence, airborne radionuclides that have been detected at the site probably originate from other sources. In order to provide a total inhalation risk estimate for current industrial receptors located on the operable unit, ambient air monitoring data are evaluated and used to provide exposure point concentrations. Air samples were collected at the northeast corner of the operable unit (Schmidt et al. 1992; air sampler location 967), from which radionuclide air concentrations were measured. These data (presented in Table 6-3) are the exposure point concentrations used to calculate the total inhalation risk for current industrial receptors. It should be noted that these data were collected after the clean soil cover was put in place, and therefore, do not include a contribution from the 200 operable unit source area.

The future scenarios are assumed not to begin until 2018. Therefore, radionuclide exposure concentrations will decay for 26 years between the time of sample collection (1992) and 2018. Half-lives and decay factors for each radioactive contaminant are provided in Table 5-4. These decay factors are multiplied by the 95% UCL soil concentrations of Table 4-20 to yield soil concentrations for the future scenarios (presented in Table 5-5). Other source depletion mechanisms (e.g., erosion, leaching, and environmental/biological degradation) are assumed not to occur.


1.0
1.1
1.25
1.4
1.6
1.8
2.0
2.2
2.5
2.8
3.2
3.6
4.0
4.5
5.0
5.6

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Figures 5-1 through 5-4 provide isopleths of future suspended particulate concentrations (in air) due to dust emissions from near surface soils and infiltration gravels/soils for each crib grouping, assuming the clean soil cover is absent. These dust concentrations are multiplied by the appropriate 95% UCL soil concentrations and the appropriate conversion factors to yield contaminant fugitive dust concentrations. An example calculation for the inhalation of fugitive dust is provided in Appendix I.

For the purpose of this risk assessment, the future industrial receptor (on the operable unit) is assumed to spend his or her working lifetime in an area of approximately 1,000,000 m^2 (250 ac). Assuming the near surface soils are uncovered and are subject to erosion, an area of this size would have an airborne particulate concentration of approximately 0.01 $\mu g/m^3$ due to the operable unit (Figures 5-1 and 5-3). Assuming that the infiltration gravels/soils are excavated and deposited on the ground surface, this same area would have an airborne particulate concentration of approximately 0.0001 $\mu g/m^3$ due to the operable unit (Figures 5-2 and 5-4). These air concentrations are multiplied by the 95% UCL soil concentrations to yield contaminant fugitive dust concentrations for the future industrial scenarios on the operable unit.

Future receptors on the Hanford Site (but outside the 200 Area) are assumed to be located along the lower, right-hand (diagonal) edge of Figures 5-1 through 5-4. Future receptors may be industrial, residential, agricultural, or recreational. Since the location of these receptors lies within the smallest numeric isopleths, the airborne particulate concentration at this point is assumed to be five times the concentration determined by these isopleths. Therefore, exposure point concentrations for receptors are calculated assuming an airborne particulate concentration of 0.005 $\mu g/m^3$ from near surface soils (Figure 5-1), and 0.00005 $\mu g/m^3$ from infiltration gravels/soils (Figure 5-2). These air concentrations are multiplied by the 95% UCL soil concentrations to yield contaminant fugitive dust concentrations for all of the future scenarios on the Hanford Site.

6.2.1.3.2 Intake Equations. Standard EPA equations, as provided in RAGS (EPA 1989a) and the HSBRAM (DOE-RL 1993b), are used as the basis for all intake calculations. Intakes of non-radioactive and radioactive contaminants are calculated and presented separately.

Non-Radioactive Contaminants. The basic equation for calculating intakes of non-radioactive contaminants via ingestion (soil or biota) or inhalation is:

$$\text{Intake} = \frac{C \times IR \times EF \times ED \times CF}{BW \times AT} \qquad \textbf{6-1}$$

where:

Intake	=	chronic daily intake of the contaminant (mg/kg-d)
C	=	contaminant concentration in the medium (e.g., mg/kg or mg/m^3)
IR	=	contact rate (e.g., mg/d or m^3/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)
CF	=	conversion factor (as appropriate)
BW	=	body weight (kg)

AT = averaging time (yr x 365 d/yr)

Intake Equation 6-1 may also be used to calculate the absorbed dose resulting from dermal exposure to contaminated soils. In this case, the contact rate is determined as follows:

$$IR_{derm} = SA \times AF \times ABS \qquad 6\text{-}2$$

where:

IR_{derm}	=	dermal exposure contact rate (mg/event)
SA	=	skin surface area available for contact (cm^2)
AF	=	soil-to-skin adherence factor (mg/cm^2-event)
ABS	=	contaminant-specific dermal absorption factor (unitless)

The dermal exposure contact rate is inserted into Equation 6-1 to yield the intake value for the dermal pathway. For the purpose of this risk assessment, it is conservatively assumed that receptors do not wear protective clothing that would limit dermal exposures.

The contaminant-specific dermal absorption factor (ABS) is a value that is either assumed or derived from the literature. Contaminants bound to a soil matrix are less dermally bioavailable than pure or dilute solutions of contaminants applied directly to skin.

Radioactive Contaminants. The quantification of exposures to radioactive contaminants requires a separate treatment. The units used to express environmental concentrations of radioactive and non-radioactive contaminants are different. Unlike non-radioactive contaminants, intake estimates for radionuclides should not be divided by body weight or averaging time. Instead, the calculated intakes represent radionuclide activities inhaled or ingested over a lifetime.

The basic equation for calculating intakes of radioactive contaminants via ingestion or inhalation is:

$$Intake = C \times IR \times EF \times ED \times CF_1 \qquad 6\text{-}3$$

where:

Intake	=	radionuclide-specific lifetime intake (pCi)
C	=	radionuclide concentration in the medium (e.g., pCi/g or pCi/m^3)
IR	=	contact rate (e.g., mg/d or m^3/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)
CF_1	=	conversion factor (as appropriate)

Intake Equation 6-3 may also be used to evaluate external exposures. In this case, the "intake" has units of pCi-yr/g, and represents the time a receptor is in close proximity to a particular radionuclide soil concentration. The "contact rate" is determined as follows:

$$IR_{ext} = ET \times RF \times CF_2 \qquad \text{6-4}$$

where:

IR_{ext}	=	external exposure contact rate (yr/d)
ET	=	exposure time (hr/d)
RF	=	dose reduction factor (unitless)
CF_2	=	conversion factor (1.14×10^{-4} yr/hr)

The external exposure contact rate is then inserted into Equation 6-3 to yield the intake value for the external exposure pathway. A dose reduction factor is used to obtain a more realistic estimate of external exposures by taking into account the effects of shielding while indoors, ground roughness, and time spent indoors.

6.2.1.3.3 Calculation of Contaminant Intakes. All exposure parameters (e.g., body weight, averaging time, contact rate, exposure frequency, and exposure duration) presented below are those recommended by the HSBRAM (DOE-RL 1993b). These exposure parameters have been specifically developed for the industrial, residential, recreational, and agricultural populations, and are used to evaluate the future exposure pathways described in Section 6.2.1.2.

Current Industrial Scenario. External exposure to radioactive contaminants is the only viable pathway from the 200-BP-1 sources under the current industrial scenario. Because of the shielding provided by the clean soil cover, calculating an intake using Equation 6-3 is not appropriate. Therefore, intake values for this exposure pathway are not calculated. For such cases, a shielding calculation can be performed by using commercially available software. For this risk assessment, the evaluation of the external exposure pathway (with consideration of the 0.5 m (1.5 ft) clean soil cover) is made using the RESRAD code (version 4.35; Argonne 1992).

The RESRAD code requires exposure parameters (i.e., shielding factor, fraction of time spent indoors and outdoors) as input for external exposure evaluations. Due to restricted access to the operable unit under current conditions, exposure parameters are different from those recommended in the HSBRAM (DOE-RL 1993b). Based on discussions with Westinghouse personnel, it is estimated for this scenario that a worker conducts survey activities on-site for 2 working days per month (i.e., 8 hr/d, 24 d/hr). In terms of RESRAD input, this is equivalent to a fraction of time spent outdoors (on site) of 0.022. A summary report of the RESRAD analysis, including input parameters, is provided in Appendix L.

Current receptors located in the 200 Area are exposed to contaminants in the ambient air that do not originate from the 200-BP-1 operable unit. Intakes of radionuclides are calculated using measurements representing air concentrations at the 200-BP-1 operable unit as well as at distant communities. These intakes (Table 6-3) are calculated by using intake Equation 6-3 and the following industrial scenario exposure parameters.

C_a	=	estimated air concentration (pCi/m^3)
IR	=	20 m^3/d
EF	=	250 d/yr
ED	=	20 yr

Future Industrial Scenario. Assuming that the clean soil cover is maintained, the external exposure pathway is also the only operative pathway under the future scenarios. The RESRAD code is used to evaluate this exposure pathway. For the future scenario, it is assumed that radiological controls no longer exist. As recommended in the HSBRAM (DOE-RL 1993b) workers are assumed to be exposed to contaminated soils for 8 hr/d, 146 d/yr. In terms of RESRAD input, this is equivalent to a fraction of time spent outdoors (on site) of 0.13. A summary report of the RESRAD analysis is presented in Appendix L.

Intake values for the future industrial scenario (year 2018), assuming contaminated soils are uncovered and subject to erosion, are presented in Tables 6-4 through 6-9. Tables 6-4 through 6-7 pertain to industrial receptors on the operable unit, while Tables 6-8 and 6-9 provide fugitive dust intakes for industrial receptors on the Hanford Site. Input parameters used to calculate these intakes are presented below.

Inhalation of Fugitive Dust. The inhalation of non-radioactive contaminants of potential concern via fugitive dust is calculated by using intake Equation 6-1 and the following industrial scenario exposure parameters. Estimated air concentrations (C_a) are the product of contaminant soil concentrations (Table 5-5), fugitive dust concentrations attributable to the operable unit (Section 6.2.1.3.1), and the appropriate conversion factor (10^{-9} kg/μg).

C_a	=	estimated air concentration (mg/m^3)
IR	=	20 m^3/d
EF	=	250 d/yr
ED	=	20 yr
BW	=	70 kg
AT	=	70 yr x 365 d/yr (carcinogens)

For the future industrial scenario, the inhalation of radioactive contaminants of potential concern via fugitive dust is calculated by using intake Equation 6-3 and the following industrial scenario exposure parameters. Estimated air concentrations (C_a) are the product of contaminant soil concentrations (Table 5-5), fugitive dust concentrations attributable to the operable unit (Section 6.2.1.3.1), and the appropriate conversion factor (10^{-6} g/μg).

C_a	=	estimated air concentration (pCi/m^3)
IR	=	20 m^3/d
EF	=	250 d/yr
ED	=	20 yr

Soil Ingestion. The ingestion of non-radioactive contaminants of potential concern in soil is calculated by using intake Equation 6-1 and the following future industrial scenario exposure parameters. Estimated soil concentrations are provided in Table 5-5.

C_s	=	estimated soil concentration (mg/kg)
IR	=	50 mg/d
EF	=	146 d/yr
ED	=	20 yr
CF	=	10^{-6} kg/mg
BW	=	70 kg
AT	=	20 yr x 365 d/yr (noncarcinogens); 70 yr x 365 d/yr (carcinogens)

The ingestion of radioactive contaminants of potential concern is calculated by using intake Equation 6-3 and the following industrial scenario exposure parameters:

C_s	=	estimated soil concentration (pCi/g)
IR	=	50 mg/d
EF	=	146 d/yr
ED	=	20 yr
CF_1	=	10^{-3} g/mg

Dermal Exposure. The dermal absorption of non-radioactive contaminants of potential concern in soil is calculated by using intake Equations 6-1 and 6-2 and the following future industrial scenario exposure parameters. Estimated soil concentrations are provided in Table 5-5.

C_s	=	estimated soil concentration (mg/kg)
SA	=	5000 cm^2
AF	=	0.2 mg/cm^2/event
ABS	=	contaminant-specific absorption factor (unitless)
EF	=	146 events/yr
ED	=	20 yr
CF	=	10^{-6} kg/mg
BW	=	70 kg
AT	=	20 yr x 365 d/yr (noncarcinogens); 70 yr x 365 d/yr (carcinogens)

Based on the conclusions of Hawkins et al. (1990), an ABS of 1% (0.01) was chosen for cadmium and nickel and an ABS of 10% (0.10) was chosen for tributyl phosphate. EPA (1992a) recommends the use of an upper bound estimate of 6% (i.e., 0.06), as an absorption factor for PCBs based on studies of 3, 3', 4, 4' tetrachlorobiphenyl. For the purposes of this risk assessment, 0.06 is used as the ABS for PCBs.

External Exposure. The intake value representing external exposure to radionuclides in soil is calculated by using intake Equations 6-3 and 6-4 and the following future industrial scenario exposure parameters. Estimated soil concentrations are provided in Table 5-5.

C_s	=	estimated soil concentration (pCi/g)
ET	=	8 hr/d
RF	=	0.8 (unitless)
CF_2	=	1.14×10^{-4} yr/hr
EF	=	146 d/yr
ED	=	20 yr

Future Residential Scenario. Assuming contaminated soils are uncovered and subject to erosion, the inhalation of fugitive dust is the only viable pathway under the future residential scenario. Intake values for the residential population are presented in Tables 6-10 and 6-11. The inhalation of non-radioactive contaminants of potential concern via fugitive dust is calculated by using intake Equation 6-1 and the following residential scenario exposure parameters. Estimated air concentrations (C_a) are the product of contaminant soil concentrations (Table 5-5), fugitive dust concentrations attributable to the operable unit (Section 6.2.1.3.1), and the appropriate conversion factor (10^{-9} kg/μg).

C_a = estimated air concentration (mg/m^3)
IR = 20 m^3/d
EF = 365 d/yr
ED = 30 yr
BW = 70 kg
AT = 70 yr x 365 d/yr (carcinogens)

The inhalation of radioactive contaminants of potential concern via fugitive dust is calculated by using intake Equation 6-3 and the following residential scenario exposure parameters. Estimated air concentrations (C_a) are the product of contaminant soil concentrations (Table 5-5), fugitive dust concentrations attributable to the operable unit (Section 6.2.1.3.1) and the appropriate conversion factor (10^{-6} g/μg).

C_a = estimated air concentration (pCi/m^3)
IR = 20 m^3/d
EF = 365 d/yr
ED = 30 yr

Future Agricultural Scenario. Assuming contaminated soils are uncovered and subject to erosion, the inhalation of fugitive dust is the only complete pathway under the future agricultural scenario. The inhalation exposure parameters and intake values for the agricultural scenario are identical to those of the residential scenario. Intake values for the agricultural population are presented in Tables 6-10 and 6-11.

Future Recreational Scenario. Assuming contaminated soils are uncovered and subject to erosion, the inhalation of fugitive dust is the only complete pathway under the future residential scenario. Intake values for the recreational population are presented in Tables 6-12 and 6-13. The inhalation of non-radioactive contaminants of potential concern via fugitive dust is calculated by using intake Equation 6-1 and the following recreational scenario exposure parameters. Estimated air concentrations (C_a) are the product of contaminant soil concentrations (Table 5-5), fugitive dust concentrations attributable to the operable unit (Section 6.2.1.3.1), and the appropriate conversion factor (10^{-9} kg/μg).

C_a = estimated air concentration (mg/m^3)
IR = 20 m^3/d
EF = 7 d/yr
ED = 30 yr
BW = 70 kg
AT = 70 yr x 365 d/yr (carcinogens)

The inhalation of radioactive contaminants of potential concern via fugitive dust is calculated by using intake Equation 6-3 and the following recreational scenario exposure parameters. Estimated air concentrations (C_a) are the product of contaminant soil concentrations (Table 5-5), fugitive dust concentrations attributable to the operable unit (Section 6.2.1.3.1) and the appropriate conversion factor (10^{-6} g/μg).

C_a = estimated air concentration (pCi/m^3)
IR = 20 m^3/d
EF = 7 d/yr
ED = 30 yr

6.2.1.3.4 Summary of Contaminant Intakes. Separate intakes are calculated for each crib grouping. Current intakes account for the clean soil cover and restricted access to the operable unit. The only current potential exposure pathway is external exposure of operable unit workers to (radioactive) contaminants of the 200-BP-1 operable unit. A means of accounting for the shielding effect of the clean soil cover has not been developed under the intake analysis utilized above. Instead, the RESRAD code (Argonne 1992) is used to conduct this analysis (Appendix L).

Receptors located in the 200 Area are currently exposed to air contaminants not originating from the 200-BP-1 operable unit. Intakes of radionuclides (Table 6-3) are calculated using measurements representing air concentrations at the 200-BP-1 operable unit as well as at distant communities.

Future intakes are calculated for each of three possibilities: the clean soil cover is maintained; near surface soils are exposed and/or excavated; the infiltration gravels/soils are exposed and deposited on the ground surface. For the future industrial scenario, it is assumed that no institutional controls are in place to limit access to contaminated soils.

The external exposure pathway is also the only complete pathway under the future industrial scenario, assuming that the clean soil cover is maintained. The RESRAD code is also used to evaluate this exposure pathway. Summary reports of RESRAD analyses are also presented in Appendix L.

Intake values for the future industrial scenario, assuming contaminated soils are uncovered and subjected to erosion, are presented in Tables 6-4 through 6-9. Tables 6-4 through 6-7 pertain to industrial receptors on the operable unit. The exposure pathways evaluated for this scenario are inhalation of fugitive dust, soil ingestion, dermal exposure, and external exposure. Tables 6-8 and 6-9 provide fugitive dust intakes for industrial receptors on the Hanford Site.

Inhalation of fugitive dust is the only pathway for which intakes are calculated for the future residential, recreational, and agricultural scenarios. Intake values for the residential scenario are presented in Tables 6-10 and 6-11. Intake values for the recreational scenario are presented in Tables 6-12 and 6-13. The intake values for the agricultural scenario are identical to those of the residential scenario, and are presented in Tables 6-10 and 6-11.

Chronic daily intakes are calculated for both noncarcinogenic effects and carcinogenic effects of non-radioactive contaminants. Intakes of radioactive contaminants are calculated only for carcinogenic effects.

6.2.2 Human Health Toxicity Assessment

The purpose of the toxicity assessment is to identify the potential adverse effects associated with exposure to site-related contaminants and to evaluate, using numerical toxicity values, the likelihood that these adverse effects may occur. The toxicity assessment for this risk assessment is conducted in accordance with RAGS (EPA 1989a) and the HSBRAM (DOE-RL 1993b).

In general, toxicity assessment is conducted in two stages: hazard identification and dose-response evaluation. Hazard identification is the determination of whether the exposure

to an agent will result in an increase in the incidence of adverse health effects, while dose-response evaluation is the process of quantitatively characterizing the relationship between the dose of a toxic substance and the corresponding incidence of deleterious effects in an exposed population (EPA 1989a). Toxicity information on chemicals and radionuclides is available in on-line databases such as the Integrated Risk Information System (IRIS; EPA 1993), Health Effects Assessment Summary Tables (HEAST; EPA 1992b), the Agency for Toxic Substances Disease Registry (ATSDR) Toxicological Profiles, and the scientific literature. The toxicological profiles for the contaminants of potential concern at this site are presented in Appendix M.

6.2.2.1 Toxicity Information for Noncarcinogenic Effects. Systemic toxic effects other than cancer can be associated with exposures to both chemicals and radionuclides. The reference dose (RfD) is the toxicity value which is used to evaluate the noncarcinogenic effects resulting from exposure to toxic chemicals. The RfD has been developed on the premise that protective mechanisms exist that must be overcome before an appreciable risk of adverse health effects is manifested during a defined exposure period. That is, there is a threshold dose which must be exceeded before adverse effects can occur. The RfD is developed to reflect the duration of exposure (e.g., subchronic and chronic exposures), and the route of exposure (i.e., inhalation and oral).

Chronic exposure is defined in RAGS (EPA 1989a) as a repeated or prolonged exposure (i.e., from seven years to a lifetime). The chronic RfD is a daily exposure level that is likely to be without an appreciable risk of deleterious effects from lifetime exposure to the general population, including sensitive subpopulations. For purposes of this risk assessment, the chronic RfD is utilized to evaluate noncarcinogenic effects which may be associated with potential exposure to the chemicals of potential concern at this site.

Carcinogens may also have systemic effects other than cancer. Carcinogens are also evaluated for potential noncarcinogenic toxic effects and are included in the determination of chronic toxicity hazard indices which characterize noncancer hazards. Carcinogenic effects, however, are usually manifested at levels that are significantly lower than those associated with systemic toxic effects; thus, cancer is usually the predominant adverse effect for contaminants that elicit carcinogenic as well as noncarcinogenic responses. For example, the threshold for deterministic effects of the most radiosensitive tissues is less than 15 rem (ICRP 1991). In contrast, EPA (56FR33050) associates a dose rate of 4 mrem/yr (lifetime dose of 0.28 rem) with a cancer mortality risk of 1×10^{-4}. As stated in RAGS (EPA 1989a), radiation exposure assessments need not consider acute toxicity effects because the quantities of radionuclides required to cause adverse effects from acute exposure are extremely large, and such levels are not normally encountered at Superfund sites.

Two chronic toxicity parameters that are used in establishing RfDs are the lowest-observed-adverse-effect levels (LOAELs) and the no-observed-adverse-effect levels (NOAELs). The LOAEL may be defined as the lowest exposure level at which there is a demonstrated statistically and/or biologically significant increase in adverse effects between the exposed animal population and the control group in a toxicological study. The NOAEL is the exposure level at which there are no demonstrated adverse effects in a dose-response toxicity study. Uncertainty factors in multiples of 10 may be further applied to the reported NOAELs or LOAELs in order to adjust for data limitations, and for differences between experimental animal exposure conditions and human exposures (National Academy of Science 1977). These factors also account for inherent variability in human responses to chemical agents, and for general imprecision in extrapolating from laboratory animals to humans.

Table 6-14 summarizes the noncarcinogenic toxicity values (i.e., RfDs) and the corresponding critical effects for the contaminants of potential concern at the site. Oral RfDs have been developed by the EPA for cadmium, nickel, and tributyl phosphate; however, there is no available noncarcinogenic toxicity factor for PCB. The detected nonradioactive contaminants at the site are not considered to be systemic toxins via the inhalation route.

Additional information on the noncarcinogenic effects for each contaminant of potential concern is presented in the toxicity profiles in Appendix M.

6.2.2.2 Toxicity Information for Carcinogenic Effects. Potential human carcinogenic effects are evaluated based on the contaminant-specific slope factor and the weight-of-evidence classification of the EPA. The weight-of-evidence classification is applied to the determination of the probability of cancer occurrence in humans, based on the strength of human epidemiological and/or animal study data. This system, originally developed by the International Agency for Research on Cancer (IARC), has been slightly modified by the EPA (1986b). Carcinogens are classified by the EPA according to the following weight-of-evidence categories:

- Group A - Human Carcinogen

  There is sufficient evidence from epidemiological studies that substantiates a causal association between exposure and carcinogenicity in humans.

- Group B1 - Probable Human Carcinogen

  There is a limited evidence of carcinogenicity in humans from available epidemiological data.

- Group B2 - Probable Human Carcinogen

  There is sufficient evidence of carcinogenicity in animals, but inadequate or no evidence in humans.

- Group C - Possible Human Carcinogen

  There is a limited evidence of carcinogenicity in animals.

- Group D - Not Classifiable

  The evidence for carcinogenicity in animals is inadequate to support classification.

- Group E - Human Noncarcinogen

  There is no evidence of carcinogenicity for humans based on adequate studies.

6.2.2.2.1 Non-Radioactive Substances. The slope factor (SF) is the toxicity value that quantitatively defines the dose-response relationship of a known or suspected carcinogen. The SF is an estimate of an upperbound lifetime probability of an individual developing

cancer following an exposure to a potential cancer causing agent. Slope factors for chemicals are expressed as the 95% upper confidence limit of the slope of the dose-response curve. The SF is derived by assuming a low-dose linearity and applying a computer model to extrapolate from the relatively high doses administered to animals (or the exposures observed in epidemiological studies) to the lower environmental exposure levels that generally occur in humans. The Carcinogen Assessment Group (CAG) of the EPA has developed upperbound SFs for carcinogens, based on the premise that there is no threshold or level of exposure below which carcinogenic effects will not be elicited.

Because the SF is the 95% upper confidence limit of the probability of a response per unit intake of a chemical over a lifetime exposure, there is only a five percent chance that the response will be greater than the estimated value. The use of SFs, thus, results in a conservative (i.e., upperbound) estimate of potential cancer risk. The true risk to humans is not likely to exceed the upperbound estimate, but may in fact be even lower. Further, because the dose-response curve is assumed to be linear in the low-dose region, the accuracy of the SF may be limited if this region should, in reality, exhibit nonlinearity.

Table 6-15 presents the carcinogenic toxicity values (i.e., slope factor) and the corresponding weight-of-evidence classifications for cadmium and PCB. Cadmium is classified as a Group B1 carcinogen, based on limited human epidemiological evidence. PCBs are classified as a Group B2 carcinogens, based on sufficient evidence of carcinogenicity in animals, but inadequate evidence of carcinogenic effects in humans. The Superfund Technical Support Center recommends a weight-of-evidence classification of Group D for tributyl phosphate (EPA 1992c). Nickel refinery dust is classified as a Group A carcinogen. However, as explained in Section 4.3.1.1, this form of nickel is not present at the site, and nickel is evaluated only for its noncarcinogenic effects.

6.2.2.2.2 Radioactive Substances. Cancer induction is the only health effect of concern resulting from exposure to environmental radioactive contamination. Systemic toxic effects occur only following relatively high doses of radiation that are not typical of environmental exposure. Currently, EPA classifies all radionuclides as Group A, human carcinogens due to their property of emitting ionizing radiation. Other low dose and low dose rate effects (such as mutagenesis, teratogenesis, and life shortening) have a quantifiable probability of occurrence, but the risk of cancer appears to be the limiting health effect (EPA 1989d).

The SFs for radionuclides are each individually determined by the EPA, based on the unique chemical, metabolic, and radiological properties of the radionuclide. Inhalation SFs are based on default values of lung clearance times for inhaled particulate radionuclides (as recommended by the International Commission on Radiological Protection [ICRP]), while ingestion SFs are based on gastrointestinal absorption fractions of radionuclides (EPA 1992b).

External exposure slope factors provide cancer risk estimates per unit exposure to a uniform radionuclide concentration in soil (i.e., an infinite slab source). Due to the conservatism of this assumption, it is not appropriate to use external exposure SFs when a significant amount of clean soil covers the contaminated materials being evaluated, as is the case at the 200-BP-1 operable unit. For such cases, a shielding calculation can be performed by using commercially available software. For this risk assessment, the evaluation of the external exposure pathway (with consideration of the clean soil cover) is made using the RESRAD code (version 4.35; Argonne 1992). RESRAD provides dose analyses (i.e., output is in mrem/yr), not cancer risk estimates. EPA risk factors (EPA 1989d) are used to convert total

doses incurred over a lifetime of exposure into risk estimates. The nominal lifetime cancer induction risk factor recommended by EPA (1989d) and HSBRAM (DOE-RL 1993b) is 6.2×10^{-4}/rem.

Dermal uptake is generally not an important route of uptake for radionuclides, which have small skin permeability coefficients (EPA 1989a). Dermal exposure to radionuclides is not evaluated in this risk assessment.

Many radionuclides have radioactive daughters that are expected to be in equilibrium with their respective parent. For this risk assessment, the radionuclide slope factors presented in Table 6-15 account for the contribution of these daughter products.

In some cases, the stable daughters of some radionuclides are inherently toxic. For example, cobalt-60 and cesium-137 decay to stable nickel and barium, respectively, both of which are systemic toxins. However, the activity concentrations of these radionuclides are associated with extremely small mass concentrations such that the toxicity of stable daughters can be ignored. For example, the 95% UCL for cesium-137 in the infiltration gravels/soils of cribs 216-B-43 through-50 (2.0E+06 pCi/g) will ultimately create only 2.3E-02 mg/kg of stable barium.

Radionuclide SFs represent best estimates (i.e., median or 50% confidence limit values) of excess cancer risk in a population per unit intake or exposure during a 70-year lifetime. As with nonradioactive carcinogens, a non-threshold dose is assumed in the evaluation of carcinogenesis related to potential exposure to radionuclides.

Table 6-15 summarizes the carcinogenicity weight-of-evidence classification and the SFs for the ingestion, inhalation, and external exposure pathways for radioactive contaminants of potential concern at this site.

6.2.2.3 Adjustment of Toxicity Factors. As stated in RAGS (EPA 1989a), for purposes of conducting risk assessment for potential dermal exposures, it is necessary to adjust an oral toxicity factor (i.e., RfD or SF) from an administered to an absorbed dose. The oral toxicity values for the nonradioactive contaminants of potential concern at this site are expressed as administered doses (i.e., intake-based). Therefore, it is necessary to adjust both the RfD or SF for these compounds in estimating potential dermal exposure to affected soils. Because dermal exposure to radionuclides is not evaluated, such an adjustment is made only for non-radioactive contaminants.

Toxicokinetic information from the available literature is generally used to determine the extent of dermal absorption for nonradioactive contaminants of potential concern. An appropriate oral absorption efficiency (expressed as percent absorbed) is identified, and the factor is applied to the RfD and/or SF to determine the corresponding dermally adjusted toxicity value. RfD values are adjusted by multiplying by the oral absorption efficiency, while SF values are adjusted by dividing by the oral absorption efficiency.

Available information in the scientific literature indicates that the oral absorption factor of PCBs is about 90%, following oral exposure in animal studies (SRC 1991b). Information regarding oral absorption of tributyl phosphate has not been determined. Therefore, an oral absorption factor of 1.0 is assumed, and the oral RfD is adopted for use as the dermal RfD without adjustment. In the case of inorganic compounds, the available information in the literature suggests that oral absorption efficiencies for these chemicals are typically in the

range of 5% to 10%, as gastrointestinal absorption is likely to be affected by several factors. Such factors include chemical form, physical state of the compound (e.g., solid or solution), particle size, dosing regimen, age, and diet. EPA (1992b) recommends a gastrointestinal absorption factor of 5% for both cadmium and nickel.

Table 6-16 presents the dermally adjusted RfDs and SFs for detected chemicals in soil at the site, including the corresponding oral absorption factors.

6.2.3 Human Health Risk Characterization

The information from the exposure assessment and the toxicity assessment is integrated to form the basis for the characterization of risks and human health hazards. The risk characterization presents quantitative and qualitative descriptions of risk.

6.2.3.1 Quantification of Noncarcinogenic Effects. Potential human health hazards associated with exposure to noncarcinogenic substances, or carcinogenic substances with systemic toxicities other than cancer, are evaluated separately from carcinogenic risks. The daily intake over a specified time period (e.g., lifetime or some shorter time period) is compared with an RfD for a similar time period (e.g., chronic RfD or subchronic RfD) to determine a ratio called the hazard quotient (HQ). Estimates of intakes for this risk assessment are based on chronic exposures. The nature of the contaminant sources and the potential for release of contaminants from the waste management units preclude short-term fluctuations in contaminant concentrations that might produce acute or subchronic effects. The formula used to estimate the HQ is:

$$HQ = \frac{\text{Chronic Daily Intake}}{\text{RfD}}$$

If the HQ exceeds unity, the possibility exists for systemic toxic effects. The HQ is not a mathematical prediction of the severity or incidence of the effects, but rather is an indication that adverse effects may occur, especially in sensitive subpopulations.

Cadmium, nickel, and tributyl phosphate are the only contaminants of potential concern which are evaluated for systemic toxicity. The remaining contaminants of potential concern (PCBs and radionuclides) are evaluated only for their carcinogenic potential.

The only reference doses that have been developed for cadmium, nickel, and tributyl phosphate are for the ingestion route of exposure. Inhalation RfDs have not been developed for these contaminants. Therefore, health effects posed by these contaminants can only be evaluated for the future industrial scenario for operable unit receptors exposed to infiltration gravels/soils through ingestion or dermal exposure.

Risk assessment summaries for future industrial scenario exposures to near surface soils are presented in Tables 6-17 and 6-18. Risk assessment summaries for the future industrial scenario exposures to infiltration gravels/soils are presented in Tables 6-19 and 6-20. These tables show that the largest HQ is 0.007. Since this value is two and a half orders of magnitude less than 1, no systemic toxic effects are expected to occur in the industrial receptor population.

6.2.3.2 Quantification of Carcinogenic Risk. For carcinogens, risks are estimates of the likelihood of an individual developing cancer over a lifetime as a result of exposure to a potential carcinogen [i.e., lifetime incremental cancer risk (ICR)]. The SF converts an intake value, as derived in the exposure assessment, to the estimated lifetime incremental risk of an individual developing cancer. The equation used to estimate cancer risk is:

$$ICR = (Intake) \times (SF)$$

Risk estimates made using this equation become increasingly inaccurate as they approach a value of 1. This is because the stochastic nature of cancer induction implies that no exposure level is high enough to ensure a carcinogenic response (i.e., ICRs must have values less than 1). For the purposes of this risk assessment, ICR values that exceed 10^{-2} are reported as "$>10^{-2}$". The ICR value calculated using the linear equation is not reported for such cases.

For non-radioactive carcinogens, intake values represent a daily intake averaged over a lifetime of exposure. Slope factors for chemical carcinogens generally represent a 95% upper confidence limit of the slope of the dose-response curve. Thus, one can be reasonably confident that the actual risk is likely to be less than that predicted. ICRs should be expressed using one significant figure only.

Intake values for radionuclides are defined to represent lifetime (not daily) exposures. Unlike most chemical slope factors, slope factors for radionuclides are best estimate, or 50% confidence limit, values.

Slope factors are specific for the route of exposure (e.g., oral SFs used to estimate risks from ingestion exposures). Because SFs for dermal exposures have not been developed, oral SFs are used as surrogate dermal SFs in accordance with RAGS (EPA 1989a). For those contaminants for which oral absorption data is available, oral SFs were adjusted to account for absorption efficiency when calculating dermal SFs.

ICRs are assumed to be additive and can be summed across pathways and contaminants. For the industrial scenario on the operable unit, ICRs are summed across all pathways (i.e., inhalation, ingestion, dermal, and external exposure) to yield a total risk associated with each crib grouping. However, the ICRs for the two crib groupings should not be summed. This is because the non-inhalation component of each of these ICRs was calculated by assuming that an industrial receptor is present at a single crib grouping for 146 d/yr, and that he would not spend his remaining time at the other crib grouping. In contrast, the ICRs for the inhalation pathway may be summed for the two crib groupings since each crib grouping contributes to the contaminant concentration to which a receptor is exposed via inhalation.

The NCP [40CFR300.430(e)(2)(i)(A)(2)] states that acceptable exposure levels represent an excess upper bound lifetime cancer risk of between 10^{-4} and 10^{-6}. The 10^{-6} risk level is considered a point of departure for determining remediation goals when ARARs are not available or are not considered sufficiently protective. Thus, cancer risks of 10^{-6} or less are considered insignificant for regulatory purposes.

6.2.3.2.1 Current Industrial Scenario. The RESRAD code (Argonne 1992) was used to calculate dose rates for current industrial receptors on the operable unit, and accounts for the shielding effects of the 0.5 m (1.5 ft) clean soil cover. Summary reports of the RESRAD

analyses are provided in Appendix L. The current dose rates for the 216-B-43 through 50 and 216-B-57 crib groupings are 0.010 mrem/yr and 0.0052 mrem/yr, respectively. Assuming a 20 yr exposure, the total dose for each crib grouping is 0.20 mrem and 0.10 mrem. Using a risk factor of 6.2×10^{-4}/rem, these dose rates are associated with cancer risks of 1×10^{-7} (216-B-43 through 50) and 6×10^{-8} (216-B-57).

Terrestrial radiation is an environmental phenomena to which all people are exposed. Woodruff and Hanf (1992) provide external radiation measurement results for distant communities (Yakima, Sunnyside, and Moses Lake) which are unaffected by the Hanford Site. The average dose rate in 1991 for these distant communities was 88 mrem/yr. Assuming this represents the naturally-occurring dose rate on the Hanford Site, workers in the 200 Area would absorb 400 mrem (8 hr/d, 250 d/yr, for 20 yr). Using a risk factor of 6.2×10^{-4}/rem, this dose rate is associated with a cancer risk of 2×10^{-4}. Thus, the industrial scenario cancer risk associated with external exposure to naturally-occurring radionuclides in the soil is more than three orders of magnitude larger than the cancer risks associated with a current industrial exposure to contaminants at the 200-BP-1 operable unit that are covered with clean soil.

Risk estimates can also be made for constituent concentrations measured in the ambient air at the 200-BP-1 operable unit and at distant off-site locations. These risks are presented in Table 6-21. Because the data upon which these risks are based were collected following the placement of the clean soil cover, these risks are associated with contaminant sources other than the 200-BP-1 operable unit (i.e., the Hanford Site, other anthropogenic sources, or natural sources). The values listed in Table 6-21 indicate that the risks associated with constituents in the ambient air at the operable unit and control locations are comparable. Operable unit workers are not exposed to excess risks via the inhalation pathway as a result of Hanford Site operations.

Table 6-22 provides a summary of the total cancer risks associated with current conditions at the 200-BP-1 operable unit, including the risks attributable to contaminants not originating from the operable unit.

6.2.3.2.2 Future Industrial Scenario (on operable unit). The RESRAD code (Argonne 1992) is used to calculate dose rates for future industrial receptors on the operable unit, assuming that the clean soil cover is maintained. Summary reports of the RESRAD analyses are provided in Appendix L. The future dose rates for the 216-B-43 through 50 and 216-B-57 crib groupings are 0.033 mrem/yr and 0.011 mrem/yr, respectively. These values are higher than the dose rates reported for the current industrial scenario. This is because the exposure frequency used to characterize the current scenario (24 d/yr) accounts for the limited access to the operable unit. The exposure frequency used for the future scenario (146 d/yr) assumes that such restrictions will no longer exist. Assuming a 20 yr exposure, the total dose for each crib grouping is 0.66 mrem and 0.22 mrem. Using a risk factor of 6.2×10^{-4}/rem, these dose rates are associated with cancer risks of 4×10^{-7} (216-B-43 through 50) and 1×10^{-7} (216-B-57). These estimates are nearly three orders of magnitude smaller than the industrial scenario cancer risk associated with naturally-occurring radionuclides in the soil (2×10^{-4}).

Cancer risks are also calculated for the possibilities that near surface soils or infiltration gravels/soils become exposed and subject to erosion. Cancer risks associated with industrial scenario exposures to near surface soils of cribs 216-B-43 through 50 and 216-B-57 are presented in Tables 6-17 and 6-18, respectively. The ICR associated with cribs 216-B-43 through 50 (9×10^{-5}) is due to external exposure to cesium-137, radium-226, and thorium-228 (and their respective daughter products). The ICR associated with crib 216-B-57 (2×10^{-5}) is due

to external exposure to radium-226 and thorium-228. The risks associated with other contaminants and pathways are less than 10^{-6}.

Cancer risks associated with industrial scenario exposures to infiltration gravels/soils of cribs 216-B-43 through 50 are presented in Table 6-19. The ICR associated with cribs 216-B-43 through 50 ($>10^{-2}$) is extremely high and is due to external exposure to cesium-137. As described in section 6.2.3.2, this ICR is reported as $>10^{-2}$ rather than the risk estimate calculated using the linear equation.

Although it is not appropriate to report the calculated risk estimate for exposure to cesium-137, this exposure can be presented in terms of radiation dose. Using a risk factor of 6.2×10^{-4}/rem, the dose associated with this exposure to cesium-137 is 7.4×10^{3} rem. Assuming an industrial exposure duration of 20 yr, this is equivalent to a dose rate of 370 rem/yr. This is nearly two orders of magnitude larger than the dose rate limit based on stochastic effects (e.g., cancer) for occupational workers at DOE sites (5 rem/yr; DOE Order 5480.11).

Strontium-90 in the infiltration gravels/soils of cribs 216-B-43 through 50 contributes a ICR of 7×10^{-3} due to soil ingestion, and cesium-137 has a soil ingestion ICR of 5×10^{-3}. The remaining notable ICRs are attributable to radium-226 (9×10^{-5} via external exposure), plutonium 239/240 (4×10^{-5} via soil ingestion), thorium-228 (1×10^{-5} via external exposure), uranium (5×10^{-6} via external exposure), and cobalt-60 (3×10^{-5} via external exposure).

Cancer risks associated with industrial scenario exposures to infiltration gravels/soils of crib 216-B-57 are presented in Table 6-20. The ICR associated with infiltration gravels/soils of crib 216-B-57 ($>10^{-2}$) is largely due to external exposure to cesium-137. Using a risk factor of 6.2×10^{-4}/rem, the radiation dose associated with this exposure to cesium-137 is 130 rem. Assuming an industrial exposure duration of 20 yr, this is equivalent to a dose rate of 6 rem/yr. This is approximately equal to the dose rate limit based on stochastic effects (e.g., cancer) for occupational workers at DOE sites (5 rem/yr; DOE Order 5480.11).

Other notable ICRs for the infiltration gravels/soils of crib 216-B-57 are attributable to radium-226 (2×10^{-4} via external exposure), cesium-137 (8×10^{-5} via soil ingestion), and thorium-228 (3×10^{-5} via external exposure).

6.2.3.2.3 Future Industrial Scenario (on Hanford Site). Cancer risks associated with future exposures to near surface soils and infiltration gravels/soils (for industrial receptors on the Hanford Site) are presented in Tables 6-23 and 6-24, respectively. The total ICR for exposure to near surface soils is 1×10^{-10}, and the total ICR for exposure to infiltration gravels/soils is 7×10^{-10}. These risk estimates represent inhalation risks due to both crib groupings.

6.2.3.2.4 Future Residential Scenario (on Hanford Site). Cancer risks associated with future exposures to near surface soils and infiltration gravels/soils (for residential receptors on the Hanford Site) are presented in Tables 6-25 and 6-26, respectively. The total ICR for exposure to near surface soils is 2×10^{-10}, and the total ICR for exposure to infiltration gravels/soils is 2×10^{-9}. These risk estimates represent inhalation risks due to both crib groupings.

6.2.3.2.5 Future Recreational Scenario (on Hanford Site). Cancer risks associated with future exposures to near surface soils and infiltration gravels/soils (for recreational receptors on the Hanford Site) are presented in Tables 6-27 and 6-28, respectively. The total

ICR for exposure to near surface soils is $5x10^{-12}$, and the total ICR for exposure to infiltration gravels/soils is $3x10^{-11}$. These risk estimates represent inhalation risks due to both crib groupings.

6.2.3.2.6 Future Agricultural Scenario (on Hanford Site). Cancer risks associated with future exposures to near surface soils and infiltration gravels/soils (for agricultural receptors on the Hanford Site) are presented in Tables 6-25 and 6-26, respectively. These risk values are identical to those calculated for the residential scenario. The total ICR for exposure to near surface soils is $2x10^{-10}$, and the total ICR for exposure to infiltration gravels/soils is $2x10^{-9}$. These risk estimates represent inhalation risks due to both crib groupings.

6.2.3.3 Evaluation of Fugitive Dust Deposition. As described in Section 6.2.1.2, inhalation is the only pathway usually evaluated for receptors who do not have contact with contaminated soils. However, due to high contaminant concentrations (in the infiltration gravels/soils) and the potential for the biota pathway to be particularly important for some contaminants, human exposures resulting from fugitive dust deposition are evaluated for a limited number of contaminants. The exposure pathways evaluated are soil ingestion, external exposure, and biota (plant, meat, and milk) ingestion.

This evaluation focuses only on those contaminants that are expected to pose the largest risks via these pathways. Contaminants are evaluated if their risk estimate from exposure on the operable unit exceeds 10^{-4}, and if the risks associated with exposure to dust deposition are expected to be greater than inhalation risk estimates (which are already evaluated for off-site locations). Tables 6-19 and 6-20 indicate that a cancer risk of 10^{-4} is exceeded only if infiltration gravels/soils are exposed and brought to the ground surface. The risk drivers for infiltration gravels/soils of crib B-43 to B-50 (Table 6-19) are cesium-137 ($>10^{-2}$) and strontium-90 ($7x10^{-3}$). The risk drivers for crib 216-B-57 (Table 6-20) are cesium-137 ($>10^{-2}$) and radium-226 ($2x10^{-4}$). However, radium-226 is an alpha emitter, and is expected to provide its greatest risk via the inhalation pathway. Since the inhalation risk attributable to radium-226 on the operable unit is extremely small ($5x10^{-12}$), radium-226 is not evaluated for fugitive dust deposition. The resulting contaminants and pathways to be evaluated for fugitive dust deposition are presented in Table 6-29.

Evaluation of these pathways is made by using the RESRAD code (Argonne 1992). RESRAD calculates radiation doses to on-site residents of radioactively contaminated sites. The code is designed to satisfy DOE Order 5400.4 requirements regarding residual radioactive materials. Although this evaluation is dose-based, the results can be converted to provide a risk-based analysis with the use of EPA risk factors.

One of the many RESRAD input parameter is the thickness of the contaminated zone. Although fugitive dust deposition is a continuous event, a single value representing the thickness of the deposited dust must be chosen. For the purpose of this evaluation, this thickness was assumed to be 0.02 mm (0.0008 in.). This is a conservative assumption compared to the total dust deposition (mass flux) values generated in section 5.2.1.3.4. The maximum mass flux deposition near the 200 Area is estimated to be $1x10^{-2}$ $\mu g/s\text{-}m^2$. Assuming the density of the deposited dust is 1.6 g/cm^3 (100 lb/ft^3), this is equivalent to a deposition of $2x10^{-4}$ mm/yr ($8x10^{-6}$ in/yr). Therefore, it would take approximately 100 yr at the calculated deposition rate for a 0.02 mm (0.0008 in.) dust thickness to accumulate. This is a conservative estimate because it assumes no resuspension of deposited dust, which would further disperse the contaminated soils.

Another RESRAD input parameter is contaminant concentration. These are determined by multiplying future contaminant soil concentrations (Table 5-6) by the mixing factor associated with the infiltration gravels/soils (Figures 5-6 and 5-8). The area of concern for this analysis is just beyond the 200 Area. According to Figure 5-6, this area is slightly beyond the 10^{-5} isopleth. Therefore, a mixing factor of $5x10^{-6}$ is used to evaluate dust deposition from infiltration gravels/soils from cribs 216-B-43 through 50. Similarly, a mixing factor of $5x10^{-6}$ is chosen to represent mixing of infiltration gravels/soils from crib 216-B-57 (Figure 5-8). The resulting contaminant concentrations attributable to cribs 216-B-43 through 50 are 5.5 pCi/g for cesium-137 and 7.0 pCi/g for strontium-90. The resulting (deposited dust) concentrations attributable to 216-B-57 are $9.4x10^{-2}$ pCi/g for cesium-137 and $1.2x10^{-4}$ pCi/g for strontium-90. These concentrations serve as the input concentrations for the RESRAD code.

Exposure parameters used to characterize soil ingestion, external exposures, and biota ingestion are those recommended by the HSBRAM (DOE-RL 1993b) for the agricultural and residential scenarios. For this assessment, it is conservatively assumed that there is no erosion of the 0.02 mm ($8x10^{-4}$ in.) thick contamination zone, and that precipitation and irrigation (which would leach away the contaminants) do not exist. RESRAD code default values were used for the remaining input parameters (e.g., food transfer factors, livestock intake, root depth).

RESRAD code output is in units of mrem/yr. To convert these values into lifetime cancer risks, the RESRAD values are multiplied by the EPA (1989d) radiation risk factor ($6.2x10^{-7}$/mrem) and the exposure duration for agricultural and residential scenarios (30 yr).

The risks associated with the various pathways following fugitive dust deposition associated with infiltration gravels/soils are presented in Table 6-30. The largest risk ($1x10^{-7}$) is due to external exposure to cesium-137 from cribs 216-B-43 through 50. Considering the conservatism inherent in the evaluation of the fugitive dust deposition, this analysis indicates that exposures resulting from fugitive dust deposition result in a total cancer risk of less than 10^{-6}.

Intake parameters for the soil ingestion, external exposure, and biota ingestion pathways are specific for each scenario. For example, the industrial scenario does not include ingestion of biota, and the industrial soil ingestion intake is only 12% of the agricultural scenario intake. Therefore, the risk estimates presented in Table 6-30 are not equally applicable to the four exposure scenarios. Exposure factors recommended in the HSBRAM (DOE-RL 1993b) are used to properly sum the risks contributed by each pathway. The resulting scenario specific risks attributable to fugitive dust deposition from cribs 216-B-43 through 50 are $9x10^{-9}$ (industrial), $1x10^{-7}$ (residential), $6x10^{-10}$ (recreational), and $1x10^{-7}$ (agricultural). The scenario specific risks attributable to fugitive dust deposition from 216-B-57 are $2x10^{-10}$ (industrial), $2x10^{-9}$ (residential), $1x10^{-11}$ (recreational), and $2x10^{-9}$ (agricultural).

6.2.3.4 Uncertainty Analysis. The risks, both non-carcinogenic and carcinogenic, presented in this assessment are not fully probabilistic estimates, but instead are deterministic estimates given multiple assumptions about exposures, toxicity, and other variables. This discussion focuses on the uncertainty surrounding the projected risks and hazards due to uncertainty in these variables.

In accordance with EPA guidance (EPA 1991a; EPA-10 1991), these variables are characterized by single point values, not probability distributions. As a result, the uncertainty

associated with estimated health impacts cannot be quantified; only a qualitative description of uncertainty is presented.

6.2.3.4.1 Uncertainty Associated with Identification of Contaminants of Potential Concern. Uncertainty in contaminant identification is related to the accuracy of the data upon which the risk assessment is based. Soil concentrations of radium-226 are highly suspect, and the risk estimates associated with this radionuclide are tentative pending further review of laboratory results. The 95% UCL for radium-226 in the infiltration gravels/soils of crib 216-B-57 was originally 380 pCi/g. However, a re-analysis of the three samples responsible for this UCL resulted in Teledyne reissuing the results as non-detects. Teledyne's review of the gamma spectra indicates that the peaks reported as radium-226 were not gamma-ray peaks but were, in fact, due to Compton interference from cesium-137. As a result the UCL for the infiltration gravels/soils of crib B-216-57 was reduced to 17 pCi/g, a value that is largely attributable to the high sample quantitation limit (80 pCi/g) of one of the three non-detects.

For this risk assessment, maximum detected concentrations for the two soil depths (near surface soils and infiltration gravels/soils) were used for comparison to background values and screening concentrations. Separate maximum detected concentrations for each of the crib groupings were not used. Therefore, it is possible for a maximum detected concentration from cribs 216-B-43 through 50 to exceed its screening concentration while the maximum detected concentration from crib 216-B-57 would not. However, the contaminant would still be retained as a contaminant of potential concern for both crib groupings. This is why the list of contaminants of potential concern for the two crib groupings (Table 4-20) are identical. The result is that contaminants are carried through the risk assessment for both crib groupings when this may not be necessary. If the contaminant identification process were performed separately for the two crib groupings, it is possible that several contaminants would no longer be considered a potential concern for one or the other crib grouping.

The process by which contaminants of potential concern are identified is designed to remove contaminants from consideration only if they pose an insignificant hazard under any scenario. Assuming that the data has been properly reported, qualified, and validated, contaminant identification is conservatively biased to cause contaminants to be carried through the risk assessment even if a more accurate treatment of the data indicates that a contaminant does not pose a risk. For example, strontium-90 was identified as a contaminant of potential concern for near surface soils. However, the highest risk associated with strontium-90 is 5×10^{-9}, due to soil ingestion of near surface soils of cribs 216-B-43 through 50 (Table 6-17).

6.2.3.4.2 Uncertainty Associated with the Exposure Assessment. Due to the potential for the 200 Area to be the site of a permanent waste management zone, only the industrial exposure scenario is evaluated for the operable unit location. This assumption implies that the operable unit would continue to be recognized as a site containing contaminated soils, and that these soils would not be exposed or excavated without appropriate controls. However, it is assumed in this risk assessment that radiological controls will not exist in 2018. In addition, evaluation of the future scenarios is divided into three possibilities: the clean soil cover is maintained and undisturbed; the near surface soils are exposed and/or excavated; the infiltration gravels/soils are exposed and deposited on the ground surface. Although the last two future settings are inconsistent with the assumption of a permanent waste management zone, they are evaluated in order to provide decision makers with an estimate of the potential human health impacts if such events are allowed to occur.

All risk estimates generated in this assessment must be interpreted with respect to the likelihood of the assumed future conditions upon which the estimates are based. For example, direct exposure to near surface soils is associated with cancer risks less than 10^{-4} (Tables 6-17 and 6-18), while direct exposure to infiltration gravels/soils is associated with cancer risks exceeding 10^{-2} (Tables 6-19 and 6-20). These risk estimates do not account for the relative likelihood for either near surface soils for infiltration gravels/soils to be exposed. Considering the depth to the infiltration gravels/soils (approximately 5 m [15 ft]), exposure of the gravels/soils is not as likely as is exposure of near surface soils. Also, neither condition is likely to occur in a permanent waste management zone. However, this information is not incorporated into the risk estimates presented in this assessment.

Because future land use of the Hanford Site is uncertain, this risk assessment evaluates four potential land uses (industrial, residential, agricultural, and recreational) of the Hanford Site (excluding the 200 Area). Each scenario is evaluated under the conservative assumption that it begins at the earliest date (2018) in which the Federal Government could release portions of the Hanford Site for non-industrial uses. The time that a scenario is assumed to be operative can be an important factor because radioactive decay can significantly reduce the concentrations to which a receptor may be exposed. For example, concentrations of cesium-137 and strontium-90 are reduced to less than ten percent of their original values within 100 yr. Since cesium-137 and strontium-90 are primary risk drivers, institutional controls for 100 yr beyond the TPA cleanup date of 2018 would reduce risk by about an order of magnitude for all pathways and land uses.

Data quality issues directly affect the exposure assessment because all intake calculations are based on exposure concentrations. Exposure concentrations are based on the 95% UCL of the mean soil concentrations. Use of the 95% UCL is intended to compensate for the uncertainty associated with choosing a single concentration to represent contaminant conditions.

The fugitive dust inhalation pathway utilizes a number of assumptions, including potential for soil erodability, soil grain-size distribution, length of each waste management unit relative to the prevailing wind, and other climatic factors. Site-specific parameter values are used when such information is available.

The assumption that contaminants are homogeneously distributed among the various soil fractions is likely to cause an underestimation of the risk associated with fugitive dust. The fugitive dust model utilized for this risk assessment assumes that the concentration of contaminants in suspended dust is equal to the soil concentration. However, many contaminants concentrate in the fine fractions of the soil which are more easily resuspended and inhaled. The result is that the risks associated with the fugitive dust pathway for many contaminants may be underestimated by an order of magnitude. Since estimated risks from the inhalation pathway on or off the operable unit are 10^{-8} or less, this uncertainty should not result in the oversight of a significant risk driver.

This non-conservative assumption is likely to be compensated for by several conservative assumptions regarding fugitive dust generation. This assessment assumes that the contaminant concentrations on the operable unit are not reduced by the emission of fugitive dust (i.e., that the unit represents an temporally infinite source). In addition, the mechanics of fugitive dust generation allows only the surface layer of erodible material to be eroded. The large soil fractions that are left behind cover the finer soil fractions beneath the ground surface, and inhibit them from being eroded. This "skin" effect is not accounted for in

this assessment. As a result, the evaluation of fugitive dust exposures is additionally conservative.

Fugitive dust deposition was evaluated by making additional conservative assumptions. Foremost among these are the assumptions that fugitive dust is not resuspended following deposition, and that precipitation (which could eventually leach away contaminants) does not occur.

Exposure parameters (i.e., body weight, averaging time, contact rate, exposure frequency, and exposure duration) represent reasonable maximum values as defined in the HSBRAM (DOE-RL 1993b), but may not reflect actual exposure conditions. For example, the future external exposure pathway uses the assumption that a worker is present 8 hr/d, 146 d/yr for 20 years. To assume that a worker is in close proximity to an operable unit for approximately half of a working lifetime, however, may not be reasonable. Consequently, such exposure condition are likely to contribute to an overestimation the risk.

6.2.3.4.3 Uncertainty Associated with the Toxicity Assessment. Uncertainty is associated with the toxicity values and toxicity information available to assess potential adverse effects. This uncertainty in the information and the lack of specific toxicity information contribute to uncertainty in the toxicity assessment.

An understanding of the degree of uncertainty associated with toxicity values is an important part of interpreting and using those values. A high degree of uncertainty in the information used to derive a toxicity value contributes to less confidence in the assessment of risk associated with exposure to a substance.

The RfDs and SFs have multiple conservative calculations built into them (i.e., factors of 10 for up to four different levels of uncertainty for RfDs, and the use of an upperbound estimate derived from the linearized multi-stage carcinogenic model for SFs) that can contribute to overestimation of actual risk. For example, Table 6-14 indicates that an uncertainty factor of 3,000 is used to calculate the oral RfD for tributyl phosphate. Table 6-15 shows that, while PCBs are evaluated as human carcinogens, the available information indicates that there is inadequate or no evidence of carcinogenicity in humans. The extrapolation of data from high-dose animal studies to low-dose human exposures may overestimate the risk in the human population because of metabolic differences, repair mechanisms, or differential susceptibility.

Although there is substantial evidence to indicate that exposure to ionizing radiation causes cancer in humans, the scenarios upon which this assumption is based are largely acute, external exposures. Sources of uncertainty specific to radionuclide exposure include: the extrapolation of risks observed in populations exposed to relatively high doses, delivered acutely, to populations receiving relatively low dose chronic exposures; estimates of doses delivered to target cells from the inhalation or ingestion of alpha-emitters (e.g., isotopes of uranium and thorium); and statistical variation in the human exposure data. In accounting for these and other sources of uncertainty, EPA risk factors for cancer incidence associated with radionuclide exposure span an order of magnitude (EPA 1989d).

EPA slope factors developed to assess external exposures to radionuclides are likely to be particularly conservative. External exposure slope factors are appropriate for a uniform contaminant distribution (that is, an infinite slab source). Because of the penetrating ability of high-energy photons, this assumption can only be satisfied if the uniform distribution of

certain radionuclides extends to nearly 2 m (6.6 ft) below ground surface, and over a distance of a few hundred meters or more. The use of the 95% UCL of the mean soil concentration to represent this uniform radionuclide concentration only compounds the conservatism inherent in the analysis of the external exposure pathway. The conservatism is expected to be worst for high-energy photon emitters such as cobalt-60 and cesium-137. The fact that the external exposure pathway is the risk driver in this risk assessment is therefore not surprising, and is more an indication of the conservatism built into the evaluation of this pathway than the actual risks associated with it.

Uncertainty is also present in the overall toxicity assessment because of: evaluation of substances that do not have toxicity values through qualitative discussion; route-to-route extrapolation of toxicity values; and potential synergistic or antagonistic interactions of substances.

PCBs have toxicity values for carcinogenic effects (i.e., SFs), but do not have toxicity values for noncarcinogenic effects (i.e., RfDs). However, exposure to PCBs may produce systemic toxic effects in addition to cancer. In addition, inhalation RfDs and/or SFs have not been developed for nickel (non-refinery dust), PCBs, and tributyl phosphate. Without an RfD or SF, quantitative evaluation of this pathway is not possible and have not been included in the overall risk estimates. However, for all contaminants of potential concern carried through the risk assessment, the level of confidence is high that key critical health effects have been evaluated.

The uncertainty associated with absorption from dermal exposure is another significant source of uncertainty that is reflected in the estimated risks associated with this pathway for some compounds. The lack of toxicity information to adequately determine RfDs and SFs for dermal exposures forces extrapolation from oral toxicity values, and compounds the uncertainty associated with the calculations. It is a common practice in risk assessment to adopt oral RfDs and SFs as the dermal toxicity values. In this risk assessment, dermal RfDs and SFs were calculated by accounting for the oral absorption efficiency. The uncertainty in this approach should be emphasized. For example, the response to an oral dose may be significantly different from the response to a dermal dose because the risk associated with point-of-entry (skin) effects for locally acting toxicants cannot be estimated from oral toxicity data. Also, dermally applied chemicals would not be subjected to "first-pass" hepatic metabolism prior to systemic circulation, as is the case for orally administered compounds. Consequently, the application of these oral dose-response relationships to dermal exposure doses is a source of considerable uncertainty in the estimated potential health risk.

6.2.3.4.4 Uncertainty Associated with the Risk Characterization. Hazard quotients and risk values provided by risk assessment by themselves do not fully characterize the health impacts associated with environmental contamination. Such a quantitative evaluation must be understood in light of the uncertainties presented above, and interpreted with respect to their significance.

Hazard quotients and cancer risks are calculated by multiplying multiple factors (e.g., contaminant concentrations, exposure parameters, toxicity values). In an effort to compensate for the uncertainty and/or natural variability in these factors, single point estimates used to characterize these factors are often conservatively biased. However, even if this bias for each factor can be considered reasonable, the product of these factors is likely to far exceed a reasonable maximum exposure. In assessing the effect of bias in the selection of parameter values, the National Council on Radiation Protection and Measurements (NCRP 1985) notes:

> ...substantial overestimation is expected when conservatism is applied in the selection of each parameter in a deterministic model. For example, in a model composed of ten or more multiplicative parameters..., the selection of only the 84th percentile for each parameter results in a predicted value that exceeds the 99.9th percentile of the distribution of model output.

This means that the risk estimates presented in a deterministic risk assessment are representative of a set of assumptions which, as a group, is extremely unlikely. Use of a more realistic set of assumptions is likely to yield significantly lower risk estimates.

Although one goal of risk assessment is to provide a hazard index and incremental cancer risk value associated with a particular scenario, doing so requires summation across pathways and multiple contaminants. The assumption of dose additivity for noncarcinogenic substances is not always appropriate because substances may have different effects in different target organs.

The summation of cancer risks also may not be appropriate because each SF, for chemical carcinogens, is an upper 95% estimate, and such probability distributions are not strictly additive. Also, summing risks from all carcinogens gives equal weight to SFs derived from animal data and SFs derived from human data.

The significance of numerical results requires interpretation. Although a 10^{-6} cancer risk may be considered insignificant, this does not imply that larger risks are necessarily significant. The NCP [40CFR300.430(e)(2)(i)(A)(2)] states that acceptable exposure levels represent an excess upper bound lifetime cancer risk of between 10^{-4} and 10^{-6}. In presenting the quantification of carcinogenic risk (Section 6.2.3.2), contaminants and pathways are described if their associated ICRs exceed 10^{-6}. However, this does not imply that ICRs greater than this value are unacceptable.

The selection of an acceptable risk level is associated with some practical considerations. For example, given the conservatism built into the evaluation of the external exposure pathway, it is desirable to measure actual does rates to confirm that the estimated risk is realistic. Using current EPA risk factors (EPA 1989d), the dose rate associated with a 10^{-4} risk is approximately 0.5 μrem/hr. Woodruff and Hanf (1992) provide external radiation dose measurement results for distant communities, which indicate that the average naturally-occurring dose rate in 1991 was approximately 10 ± 0.3 μrem/hr. Even if a 10^{-4} cancer risk due to external exposure were present at the 200 Area, the associated dose rate would be smaller than the fluctuation in background levels, and could not be measured. The implication is that it is impossible to use dose measurements to disprove the existence of an external exposure cancer risk of 10^{-4} or less. Since current risk estimates due to external exposure do not exceed 10^{-7} (Table 6-22), environmental dose measurements are of very limited usefulness.

6.2.3.5 Human Health Risk Characterization Summary. This baseline risk assessment evaluates the human health risks posed by contaminants in the 200-BP-1 operable unit under four exposure scenarios (industrial, residential, recreational, and agricultural) and three locations (operable unit, Hanford Site, off Hanford Site). This evaluation is performed for current conditions as well as several future conditions. Table 6-1 indicates the locations and times for which each of the exposure scenarios is evaluated.

Non-radioactive contaminants were evaluated for both non-carcinogenic and carcinogenic effects, as appropriate. Radioactive contaminants were evaluated only for their carcinogenic potential.

The largest estimated HQ is 0.007 (associated with infiltration gravels/soils of cribs 216-B-43 through 50). Since this value is two and a half orders of magnitude less than 1, no systemic toxic effects are expected to occur as a result of exposure to contaminants at the operable unit. The remaining summary focuses on estimates of cancer risk.

A summary of ICRs associated with non-radioactive and radioactive contaminants of potential concern under current conditions is provided in Table 6-31. Evaluation of the current scenario recognizes that a clean soil cover currently exists at the operable unit (Hayward 1992), and that access to the unit is currently limited. Therefore, the only operative exposure pathway is external exposure to workers on the operable unit. For both crib groupings, the estimated cancer risks are less than 10^{-6}.

Environmental monitoring data can be used to estimate cancer risks associated with contaminants attributable to sources other than the 200-BP-1 operable unit. Air monitoring data from the 200 Area represent ambient air concentrations of constituents to which workers on the operable unit are exposed. External radiation data from distant communities represent naturally-occurring sources and general anthropogenic sources (e.g., worldwide fallout from atmospheric nuclear bomb testing) of external exposure to which operable unit workers are exposed. The risks associated with these exposures (presented in Table 6-22) indicate that the total risk (2×10^{-4}) incurred by workers on the operable unit is dominated by external exposure to naturally-occurring radionuclides in the soil, and that the risk associated with operable unit contaminants is three orders of magnitude smaller than this total.

A summary of ICRs associated with non-radioactive and radioactive contaminants of potential concern under future conditions is provided in Table 6-32. Evaluation of the future conditions is divided into three possibilities:

- The clean soil cover is maintained and undisturbed
- The near surface soils are exposed and/or excavated
- The infiltration gravels/soils are exposed and deposited on the ground surface.

Although future land use of the 200 Area is assumed to be limited to industrial activities, for the purposes of this baseline risk assessment radiological controls are assumed to be nonexistent for the future industrial scenarios. Even without restricting access to the operable unit, the ICR associated with the industrial scenario on the operable unit is less than 10^{-6} as long as the clean soil cover is maintained.

Assuming near surface soils are exposed and subject to erosion, the largest risk for operable unit receptors is 9×10^{-5}, associated with cribs 216-B-43 through 50. Risk estimates for receptors on the Hanford Site (excluding the 200 Area) represent inhalation exposures, and are all less than 10^{-6}.

Although infiltration gravels/soils are approximately 5 m (15 ft) below ground surface, the human health impact associated with excavation of these materials is also evaluated. Risk estimates for industrial receptors on the operable unit exceed 10^{-2} for both crib groupings.

Because risk estimates made using a linear dose-response equation become increasingly inaccurate as they approach a value of 1, ICR values that exceed 10^{-2} are reported as $>10^{-2}$.

Because of the high risks associated with exposure to infiltration gravels/soils for operable unit receptors, risk estimates for Hanford Site receptors were calculated for exposures resulting from fugitive dust deposition of cesium-137 and strontium-90. These estimates were added to the risk estimates associated with inhalation exposures. Although the evaluation of fugitive dust deposition in this assessment was conservatively biased, the total ICRs for all scenarios on the Hanford Site are less than 10^{-6}.

In summary, this risk assessment indicates that a cancer risk of 10^{-6} is exceeded only under the future industrial scenario for receptors on the operable unit, and only if near surface soils or infiltration gravels/soils are uncovered, permitting direct contact with contaminants. These risk estimates do not consider the probability that the clean soil cover will be absent or present in the year 2018, or that future operable unit workers will excavate the infiltration gravels/soils. In addition, because this is a deterministic risk assessment, the uncertainty associated with these risk estimates cannot be quantified. In order to compensate for the uncertainty associated with input parameters, estimates used to characterize these parameters are often conservatively biased. As a result, the risk estimates provided in this assessment represent a set of assumptions which, as a whole, is extremely unlikely. Use of a more realistic set of assumptions is likely to yield significantly lower risk estimates.

6.3 ECOLOGICAL RISK ASSESSMENT

The purpose of the ecological risk assessment is to estimate the potential present and future baseline ecological risks for the 200-BP-1 operable unit contaminants to ecological receptors. These receptors include all organisms, except humans and domestic animals, potentially exposed to site contaminants. The 200-BP-1 operable unit is a source operable unit and only terrestrial organisms are considered in the baseline risk assessment.

Because it is not possible to evaluate all potential effects on all potential receptors, this assessment focuses on so called "indicator species" or receptor to represent a variety of exposure pathways and trophic positions. Assessments combined modeling data, soil data, and other supportive information to evaluate exposure of receptor species to both inorganic and radiological constituents.

6.3.1 Problem Formulation

Issues relevant to evaluating ecological risk at the 200-BP-1 operable unit are:

- Identifying the contaminants of concern that occur in concentrations greater than background
- Identifying the media in which these contaminants occur
- Characterizing the primary pathways of biological exposure
- Identifying biological resources in and near the operable unit
- Identifying key biological receptors and their natural histories
- Defining a conceptual ecological model for key receptors
- Obtaining transfer coefficients for contaminant movement between elements of the conceptual model

- Estimating doses, daily intake rates, or body burdens from all contaminants based on transfer coefficients, the conceptual model, fraction of the receptor's habitat that is contaminated, and exposure duration and frequency
- Comparing doses, daily intake rates, or body burdens to established benchmarks
- Estimating incremental individual and population risk from the 200-BP-1 Operable Unit.

6.3.1.1 Stressor Characteristics. From Section 4.4 the stressors of concern identified in near surface soils are Cs-137, Ra-226, Sr-90, Th-228, and total U.

Cesium-137 is a beta and gamma emitter. Radium-226 is primarily an alpha emitter, but also emits gamma radiation. Likewise, thorium 228 is primarily an alpha emitter, but also emits gamma radiation. Uranium-234 and uranium-235 emit alpha and gamma radiation, and uranium-238 is an alpha emitter. Strontium-90 is a beta emitter, and the decay product Y-90 is a beta emitter. Half-lifes are given in Table 5-4. A brief summary of the properties of these contaminants is provided in Section 5.1.

6.3.1.2 Ecosystem/Components Potentially at Risk. Contaminants found in the 200-BP-1 operable unit near surface soils are all radioactive elements. Radioactive elements have ecological effects resulting from their presence in the abiotic environment (external dose), from ingestion (e.g., dose from food consumption), and from accumulation of the element in the body (dose from body burden). Total daily doses to an organism can be estimated as the sum of doses (weighted by energy of radiation) received from all radioactive elements ingested, inhaled, residing in the body, and available in the organism's environment. Radiological dose calculation methodology has been reviewed by Baker and Soldat (1992), and were applied in this risk assessment.

All contaminant elements have been found in the soil within the operable unit. No analyses have been conducted to evaluate the extent of contamination outside the 200-BP-1 operable unit fence. The operable unit does not contain surface water bodies and is not apparently subject to mass flows from surface water runoff. No data have been obtained to evaluate concentrations of contaminants in biological media within the operable unit; consequently, biological uptake must be evaluated from a soil source term. Transfer of contaminants to plants was evaluated via bioconcentration factors or transfer ratios from soil to plants. Animal uptake on the basis of ingestion of contaminants in food stuffs was evaluated using both transfer ratios and gastro-intestinal absorption efficiencies, biological half-lives, and food intake rates; ingestion and inhalation of soils have been neglected because of the lack of information available to quantify these pathways. In this regard, dose estimates were not conservative.

Components of the 200-BP-1 operable unit environment that may be affected by wastes at the site are indicated by the vegetative communities present at the operable unit and its environs. These communities have not been defined during the RI on the basis of site-specific investigations, and studies have been conducted to evaluate the wildlife species using the site. Other studies of the 200 Areas and the 200 Area plateau conducted over the past 20 years have examined plant communities and wildlife associations to provide indications of the ecosystem potentially affected by the 200-BP-1 operable unit (see Section 3.7.2).

6.3.1.3 Ecological Effects. Ionization radiation can impact wildlife depending upon the level of exposure. Exposure can be either acute or chronic. Depending upon the concentration of exposure, acute exposures can result in organism mortality, generally characterized as the LD-50 (concentration to cause 50% mortality in some specified period of time). For mammals this is usually 30 days. Other possible effects from acute exposure are physiological and pathological changes, developmental and reproductive effects. Effects from chronic exposure include physiological, reproductive, growth, and development effects.

The radiological dose an organism receives is usually expressed as rad/day. Exposure can result from both external environmental radiation and internal radiation from body burden. Both exposure pathways are added in determining total organism dose. Internal exposure must include both body burden and foodchain uptake.

Unfortunately, most of the available information concerning ionization radiation is for acute high dose and not for low exposure and chronic effects (Rose 1992). The use of acute data extrapolated to chronic levels is not in all cases appropriate and must be viewed with caution. For example, during chronic exposure there is a point where competition between injury and natural organism repair mechanisms are balanced resulting in no effect (Ophel et al. 1976).

The major area of uncertainty in evaluating ecological effects from exposure to ionizing radiation is extrapolation of the individual to the population level of ecological organization.

6.3.1.4 Endpoint Selection. The assessment endpoint for study is the health of selected receptor organisms and their populations and the measured endpoint is individual mortality. However, there were no mortality studies conducted on indicator species. The only endpoints used in the ecological risk assessment that were directly measured were the chemical analysis of near-surface soils. The focus of this study is at the individual level of ecological organization using several trophic levels of the terrestrial foodchain in the 200 BP-1 Operable Unit. The use of several trophic levels encompasses organisms of varying sensitivity and several contaminant transport pathways.

6.3.1.5 Conceptual Model. Based on the descriptions of ecological resources present at, or near, the 200-BP-1 operable unit and assuming a contaminant source limited to the soil, a conceptual ecological model can be derived for the key ecological resources of the area (Figure 6-1). In this model, uptake of contaminants from soil by vegetation serves as the basic source of contaminant entry into the food chain. The herbivore component, represented in the model by insects, the dominant herbivorous mammals, and the dominant herbivorous (seed-eating) bird, acts as the primary conduit between contaminants in vegetation and contaminants in carnivores. Deer, although present on the 200 Area plateau, are not common within the 200 Areas because of the high barb-wire-topped fence surrounding the area. Consequently, this receptor was neglected in the conceptual model. Two levels of carnivores are common to the 200 Area plateau and the Operable Unit: the primary carnivores prey almost entirely on herbivores, consequently only three levels of bioaccumulation are possible (soil to plant, plant to herbivore, herbivore to primary carnivore). Second-order carnivores prey on other carnivores as well as on herbivores. Bioaccumulation in these animals is more complex and was evaluated using information on dietary composition. Key receptors evaluated in this risk assessment were Great Basin pocket mice, black-tailed Jack rabbits, Swainson's hawks, burrowing owls, coyotes, and loggerhead shrikes.

Estimating ecological risks from contamination in the 200-BP-1 operable unit becomes problematic when considering animals whose habitat use extends beyond the operable unit boundary. The 200-BP-1 operable unit is a relatively small area within a much larger zone of contamination in the 200 East Area. However, the operable unit also is located on the north edge of the 200 East Area within 100 m (330 ft) of near-pristine sagebrush/Sandberg's bluegrass habitat. Contaminant concentrations in this habitat are expected to be much lower than within the 200 East Area due to the lack or unimportance of distribution pathways from the 200 East Area into this region. The area lies generally upwind of the operable unit, so wind-blown contamination is likely to be insignificant. Surface water flows from rainfall are rare and the operable unit has never been used as an impoundment for liquid waste discharge, therefore the potential for water-born movement of contaminants laterally from the operable unit is also likely to be insignificant. Consequently, the environment outside the 200-BP-1 operable unit as used by most of the wide-ranging animals in the conceptual model is likely to be a mix of contaminated and uncontaminated habitat.

Because the operable unit is small relative to the home ranges of animals such as hawks, owls, loggerhead shrikes, and coyotes, the incremental risk to these resources from the operable unit is likely to be small. This incremental risk may be insignificant if an uncontaminated environment outside the operable unit is assumed, but such an assumption is insupportable and would inordinately lower the estimation of risk significance. A worst case assumption would be that the contaminant environment of the receptor outside the operable unit is not much different from that within the 200-BP-1 operable unit environment. Such an assumption would be highly conservative for the larger raptors and the coyote, who range over many km^2. A reasonable estimation of risk for these receptors lies somewhere between these extremes.

6.3.2 Analysis

The analysis phase of the ecological risk assessment is a technical evaluation of the available data to assess the potential effects of exposure to the stressors on the target receptors previously discussed.

6.3.2.1 Characterization of Exposure. The available analytical data for the 200-BP-1 operable unit is limited to surface soil samples (surface is defined as 0-4.6 m [0-15 ft] depth). The section focuses on developing the exposure relationship between receptors and site contaminants. For the purpose of the exposure characterization, the 95% upper confidence values of the data set was used to establish the exposure scenario concentration. These concentrations are shown in Table 4-20. UCL values are developed for both crib groupings; the higher of the two values is used as the contaminant concentration in this assessment. It was assumed these concentrations were uniformly distributed over the site and were biological active and available for transport into the biosphere. It was also assumed that the measured concentrations for the radionuclides were the concentrations appropriate at the time of the risk assessment and no decay was considered.

6.3.2.2 Stressor Characterization. For the purpose of this risk assessment, the soil concentrations were assumed to be uniformly distributed across the soil surface of 200-BP-1 operable unit.

6.3.2.3 Ecosystem Characterization. A detailed characterization of the ecosystem of the 200-BP-1 operable unit was previously given in Section 3.7.2. This section discusses the point of contact between the receptor organisms and the stressors.

The spatial distribution or the home range of target organisms was evaluated from available site data to establish the point of contact (length of exposure to COCs) between the stressor and receptor organisms. The overlap of receptor home range with the site was considered sufficient for evaluation as a potential receptor and it was assumed that at least part of its life is spent within the site. The period of exposure of an organism was determined by evaluating the percentage of time an organism could potentially spend feeding within the site. This was determined by estimating the fraction of the site area within the receptor home range area. No attempt was made to discriminate between seasonal use of the site by receptors, because site contaminant concentrations were assumed to remain constant and not vary seasonally.

For organisms whose home range is smaller than the operable unit, it was assumed that 100% of their diet consisted of contaminated foodstuffs. However, for organisms spending a fraction of their time feeding within the operable unit, a usage factor was calculated based on the proportion of their home range that the operable unit could encompass. For example, for a Swainson hawk, whose home range is about 60 times larger than the 200-BP-1 operable unit, an usage factor was calculated by dividing the area of the operable unit (0.101 km^2) by the hawk's home range (5.77 km^2). This usage factor was incorporated into the dose equation. In some cases a home range was derived from species densities observed on the Hanford Site. This is a reasonable derivation if the species of interest is territorial such as the loggerhead shrike and burrowing owl.

6.3.2.4 Exposure Analysis. The purpose of the exposure analysis is to integrate the spatial and temporal distributions of the ecological components and stressor to evaluate exposure.

As previously discussed, all constituents identified as potential concern in the human health risk assessment before screening of constituents of greatest human health risk were considered to be of concern in the ecological risk assessment. Because of the lack of site-specific data other than soil, it was assumed all receptors spend some fraction of their life in the 200-BP-1 operable unit, and obtain all their food from the site when present, and all consumed food is contaminated. However, because there is no source of water within the site, drinking water was not considered a route of exposure. Because of the lack of site-specific data for plants and wildlife, this risk analysis can only be considered a screening-level analysis.

For radiological constituents, estimated plant tissue and wildlife concentrations were converted to dose. Total dose for all radionuclides was compared to published effects levels and regulatory standards where available. Total organism dose was obtained by adding dose from each nuclide. The total radiological dose was also compared to some benchmark value.

6.3.2.5 Exposure Profile. The ecological risk assessment focuses on potential noncarcinogenic effects on vegetation and wildlife potentially exposed to constituents present in the 200-BP-1 operable unit. Terrestrial vegetation is represented as a generic plant species for uptake from the soil and as a food source for wildlife. Terrestrial wildlife species evaluated were selected based upon their presence at the site, trophic position, and habitat requirements. For the purpose of the ecological risk assessment, the 95% upper confidence limit was used as the exposure concentration for modeling and dose calculations.

The major route of contaminants to plants was assumed to be direct uptake from soil. Ingestion was assumed to be major route of exposure to wildlife species for both nonradiological and radiological constituents. For radionuclides the exposure pathway considered uptake from contaminated food resulting in internal exposure. Table 6-33 gives exposure pathways for the ecological receptors. Depending upon the receptor, several potential foodchain exposure pathways are presented. For both radiological and nonradiological contaminants, the dose was based on receptor whole-body concentrations.

Intake of Constituents of Concern by Terrestrial Receptors

This section describes the methods used to estimate intake of constituents of concern in surface soils by terrestrial organisms for the 200-BP-1 operable unit. A summary of exposure parameter for each study organism is given in Table 6-34. Shown are typical receptor weights, food ingestion rates, and effective radius (radius of the receptor used in the radiological dose calculations). Dose from radionuclides was calculated based upon the computer code developed by Baker and Soldat (1992). For a complete description of the code, see Baker and Soldat (1992).

Radionuclide concentrations in primary organisms can be calculated directly from soil concentrations and transfer factors. The primary organism is a plant. Radionuclide concentrations for secondary organisms can be calculated from their diet of primary organisms. Representative secondary organisms are birds and mammals. The transfer coefficients from soil to plants and soil to insects are given in Table 6-35. For insects, an arbitrary transfer coefficient of 1:1 was used. The accuracy of this value is not known because no information relating transfer of study constituents from soil to insects was found. This estimate has a high uncertainty.

The internal total-body dose rate to an organism for N radionuclides is given as

$$R_c = \sum_{i=1}^{N} b_{i,c} E_{i,c} \qquad \text{6-5}$$

where

R_c = dose rate to total body of organism c (rad d^{-1}),
$E_{i,c}$ = effective absorbed energy rate for nuclide i per unit activity in organism c (Rad Ci^{-1} d^{-1}).
$E_{i,c}$ = $\epsilon_{i,c}$MeV dis^{-1} x 3.70E10 dis s^{-1} Ci^{-1}

86,400 sd^{-1} x 1.602E-11 Rad^{-1} MeV = 5.12E4 $\epsilon_{i,c}$

where

ϵ is the effective absorbed energy for nuclide i in organism c.
$b_{i,c}$ = specific body burden of nuclide i in organism c (Bq kg^{-1}).

For a primary organism,

$$b_{i,c} = C_{i,c} B_{i,c} \quad 6\text{-}6$$

where $C_{i,c}$ is concentration of nuclide i in soil to which organism c is exposed (Bq m^{-3}), and $B_{i,c}$ is bioaccumulation factor for nuclide i and organism c (m^3 kg^{-1}).

Combining equations 6-5 and 6-6 yields the dose rate in rad d^{-1} to the primary organism

$$R_c = \sum_{i=1}^{N} C_{i,c} B_{i,c} e_{i,c} \quad 6\text{-}7$$

For the secondary organism, we can write an expression for a single radionuclide equating the change in body burden for the uptake and removal of the radionuclide

$$\frac{db^s}{dt} = \frac{P}{M} - \lambda b^s \quad 6\text{-}8$$

where

- b^s = specific body burden of the secondary organism (Ci/kg)
- P = rate of uptake of radionuclide by body of organism (Ci/d)
- λ = $(\lambda_b + \lambda_r)$ effective decay constant in secondary organism, (d^{-1}), where $\lambda_b = \ln(2)/T_b$ is the biological removal rate constant for the nuclide in the secondary organism and $\lambda_r = \ln(2)/T_r$ is the radiological decay constant for the nuclide
- M = mass of secondary organism (kg).

The secondary organism uptake rate is given by

$$P = b U f_1 \quad 6\text{-}9$$

where

- b = body burden of primary organism (Ci/kg)
- U = intake rate of primary organism by predator (kg/d)
- f_1 = fraction of radionuclide initially retained in total body of secondary organism (unitless).

Solving equation 6-8 with $b^s = 0$ when $t = 0$:

$$b^s = \frac{P}{M}\frac{(1-e^{-\lambda T_s})}{\lambda} \quad 6\text{-}10$$

where T_e is the period of exposure (d).

Then, for a secondary organism c, the dose rate in terms of the body burden of the primary organism for N radionuclides is

$$R_c = \sum_{i=1}^{N} \frac{B_i U_c f_{1,i}}{M_c} \frac{(1-e^{-\lambda_{i,c} T_e})}{\lambda_{i,c}} E_{i,c} \qquad 6\text{-}11$$

where

U_c = intake rate of primary organism by secondary organism c (kg/d)
$\lambda_{i,c}$ = effective decay constant of nuclide i in secondary organism c (d^{-1})
M_c = mass of secondary organism c (kg).

In the absence of specific data, the removal constants, $\lambda_{i,c}$, and uptake fractions, $f_{1,i}$, are taken to be that of Standard Man as derived from Publication 2 of the International Commission on Radiological Protection (ICRP 1959). The values of effective energy, $\epsilon_{i,c}$, were determined knowing the effective radius of the organism. The exposure time, T_e, is usually assumed to be 1 year for regulatory purposes, and the concentration is averaged over 1 year. These doses to organisms may be obtained by hand calculation or the CRITR2 screening program. External dose to wildlife from radionuclides was not calculated because it has been show to be a minor contributor to dose (Poston and Soldat 1992)

Species-specific intake rates were given in Table 6-34. The fraction ingested from a contaminated source was based on the animal's home range and the amount of food expected to be consumed from contaminated areas. Feeding rates are typically reported on a wet-weight basis, while contaminant concentrations in soil and biota are reported on a dry-weight basis. Dry-weight to wet-weight conversion factors are given in Table 6-36.

Plants

The 95% upper concentrations for each constituent in soil were shown in Table 4-20. The concentration factors from soil to the generic plant was obtained from available literature (Table 6-37). The maximum reported transfer coefficients from soil to plants were used in all dose calculations. These values were used to model plants as a food source in successive trophic levels.

Wildlife

Species-specific ingestion rates and body weights were presented in Table 6-34.

To evaluate potential exposure of Swainson's hawk, burrowing owl, loggerhead shrike, Great Basin Pocket mouse, Jack rabbit, and coyote from soil contaminants, transfer coefficients were used. Transfer of contaminants from ingestion of prey species were either estimated from available literature or from plant to beef transfer coefficients. If site-specific data were not available for transfer from plants to mouse, plant to beef values were used.

Calculated dose to wildlife species for radionuclides are given in Tables 6-38 to 6-43. The species to receive the highest potential radiological dose was the loggerhead shrike, which received a total dose of 5.80 rad/d for a strict diet of small mammals (Table 6-41). A total of 96% of the dose was received from Sr-90. However, because the shrike eats both small mammals and insects the dose received from eating a strict diet of insects is also shown. This dose was 0.012 rad/d. The exact contribution of each food source to the shrike's diet is not known and will be somewhere between these two values. Likewise, dose to the burrowing owl for a small mammal diet was 2.94 rad/d, and the dose from a food source of 100% insects was 0.00654 rad/d. Again, the more realistic dose lies somewhere between the two values; however, the more an animal feeds on mice, the higher the modeled dose.

The species that received the third highest dose was the pocket mouse, which received 0.159 rad/d. The order of decreasing dose was Swainson's hawk (0.110 and 0.000272 rad/day calculated for a diet of mice and insects, respectively), Jack rabbit (0.0678 rad/d from a plant diet), and coyote (0.001 and 0.102 rad/d for vegetation and mice diets, respectively). The dose from radionuclides or antimony to the mule deer was not calculated because the 200 Areas' fence will exclude deer from the operable unit. For all organisms, Sr-90 was the major contributor to radiological dose.

6.3.2.6 Characterization of Ecological Effects. The purpose of this section is to analyze the relationship between the stressor and assessment and measurement endpoints.

Evaluation of Relevant Effects Data

The only regulatory driver for radionuclides in the environment is DOE Order 5400.5, which requires exposure to aquatic organisms to be less than 1 rad/day. Because of the lack of radionuclide data for terrestrial organisms, this limit was also applied to terrestrial organisms in the 200-BP-1 operable unit. Other toxicity data were also evaluated.

The most recent and perhaps one of the most inclusive reviews on the effects of ionizing radiation on terrestrial organisms was completed by Rose (1992). Rose summarized the sensitivities of wildlife to ionizing radiation; this work was used as a major reference.

Rose (1992) reported the lower limits of lethal effects for chronic irradiation was 360 rad/yr or roughly 1 rad/d for several American rodents. The lower limit for red pine was reported to be around 0.82 to 1.64 rad/d for continuous exposure (Sparrow et al. 1963, 1970). Semagin (1975) reported a dose of 0.008 rad/d as the lowest dose that produced an effect on the fetuses of laboratory rats irradiated during the third period of intrauterine life. It was found that body mass was reduced and brain mass increased at birth. The increase in brain mass was the result of nerve tissue and not oedema. The reported range for developmental and behavioral changes from chronic irradiation exposure was also summarized by Rose (1992). An exposure of 0.49 rad/d did not effect the growth rate of several American rodents, e.g., *Peromyscus leucopus* (Childs et al. 1966). Pocket mice (*Pergnathus formosus*) were reported unaffected at a dose of 0.96 rad/d. Mellinger and Schultz (1975) reviewed the literature on the effects of ionizing radiation on birds. The LD-50 ranged from 460 to 3,000 rad.

In another extensive review of the affects of ionizing radiation on terrestrial organisms, the International Atomic Energy Agency (IAEA 1992), concluded that a "dose rate of approximately 10 mGy/d (1 rad/d) represents the threshold at which slight effects of radiation become apparent in those attributes, e.g., reproduction capacity, which are of importance for

the maintenance of the population." IAEA concluded that "reproduction was the population attribute most sensitive to damage from chronic irradiation and also the attribute of greatest significance in the ecological context".

6.3.2.7 Ecological Response Analysis. The purpose of this section is to analyze data used to characterize ecological effects.

Stressor-Response Profile

The highest radiological dose to loggerhead shrikes and burrowing owl exceeded the DOE Order of 1 rad/d. These values also exceeded the threshold of 1 rad/d to protect against reproductive effects, recommended by IAEA (1992). No species specific toxicity data were found for these organisms.

The highest radiological dose to the pocket mouse was 0.159 which is below the IAEA recommendation and DOE Order of 1 rad/d and below any reported effects on the pocket mouse (French et. al., 1966). However, Semagin (1975) reported that a dose of 0.008 rad/d effected the fetuses of laboratory rats irradiated during the third period of intrauterine life by reduction of body mass and a increase brain mass.

The highest estimated dose to the jack rabbit, Swainson's hawk, and coyote was 0.0678, 0.110, and 0.102 rad/d, respectively. These values are well below the DOE Order and the recommendation of the IAEA for exposure to ionizing radiation.

6.3.3 Risk Characterization

The risk characterization phase attempted to evaluate the likelihood of an adverse effect to receptor organisms potentially found in the 200-BP-1 operable unit.

6.3.3.1 Risk Estimation. The likelihood of an adverse effect to receptor species was estimated through a hazard index. The hazard index is defined as the ratio of the contaminant dose to some benchmark dose/concentration, i.e., DOE Order.

$$HI = \frac{\text{Organism's Dose}}{\text{Benchmark Dose}}$$

The HI ratio is used to assess the potential adverse effect to an individual. For example, an HI that approaches or exceeds unity would strongly indicate an adverse effect to an individual. However, for this risk assessment a value approaching 20% of the benchmark dose (0.2 rad/d) will be used as an indicator of potential adverse effects. The 0.2 rad/d was used to provide adequate protection of the receptor population and allow for the assessment and endpoint evaluation. A 20% variation in a population is approximately the limit of detection of field measurements considering habitat variations, species differences, and sampling methods (Suter et al. 1992).

A summary of calculated HIs for the receptor organisms is shown in Table 6-44. Using the benchmark of 0.2 rad/d, the loggerhead shrike, and the burrowing owl could be adversely affected from exposure to ionizing radiation.

6.3.3.2 Integration of Stressor-Response and Exposure Profiles. The purpose of this section is the integration of receptor dose values for the constituents of concern with expected biological responses and describe the significance of risk to the various ecological receptors.

6.3.3.2.1 Uncertainty. The uncertainty associated with the approach used in the ecological risk assessment for the 200-BP-1 operable unit is significant because data only exist for constituents of concern in soil. Use of the 95% upper concentration level is a highly conservative assumption. The ecological risk assessment was generally modeled. Modeling from soil to potential ecological receptors required a number of assumptions including soil-to-plant transfer factors or coefficients. A review of the literature produced a range of values. Again, to take the conservative approach, in all cases the highest transfer factor was used. Other assumptions included estimating the time that a receptor spends feeding within the unit and exceeds unity would strongly indicate an adverse effect to an individual. However, for this risk assessment a value approaching 20% of the benchmark dose (0.2 rad/d) will be used as an indicator of potential adverse effects. The 0.2 rad/d was used to provide adequate protection of the receptor population and allow for the assessment and endpoint evaluation. A 20% variation in a population is approximately the limit of detection of field measurements considering habitat variations, species differences, and sampling methods (Suter et al. 1992). The HI at or above 1 would indicate an potential measurable risk.

A summary of calculated HIs for the receptor organisms is shown in Table 6-44. Using the benchmark of 0.2 rad/d, the loggerhead shrike, and the burrowing owl could be adversely affected from exposure to ionizing radiation.

6.3.3.2.2 Risk Description.

Ecological Risk Summary

The results of the risk assessment do suggest a potential impact to loggerhead shrike and burrowing owls from exposure to ionizing radiation. These results do suggest additional studies are necessary to confirm this risk.

Interpretation of Ecological Significance

The approach presented for the 200-BP-1 operable unit evaluated several important and relevant pathways for contaminant movement within the 200-BP-1 operable unit and the most likely ecological receptors to show an intercept for these pathways and demonstrate an impact. The approach taken modeled contaminants of potential concern uptake from soil/plant to receptors at several key trophic positions and ecological relevance to the Hanford Site. The extrapolation of the identified risk from the measured endpoint of mortality to the assessment endpoint of population health does suggest the likelihood of a risk to the loggerhead shrike and burrowing owl populations. However, estimates of risk will probably overestimate real risk because of the conservative exposure scenario employed. The HI indicates that verification of risk conclusions would be warranted in light of the limited data set and conservative assumptions on which this risk assessment was conducted.

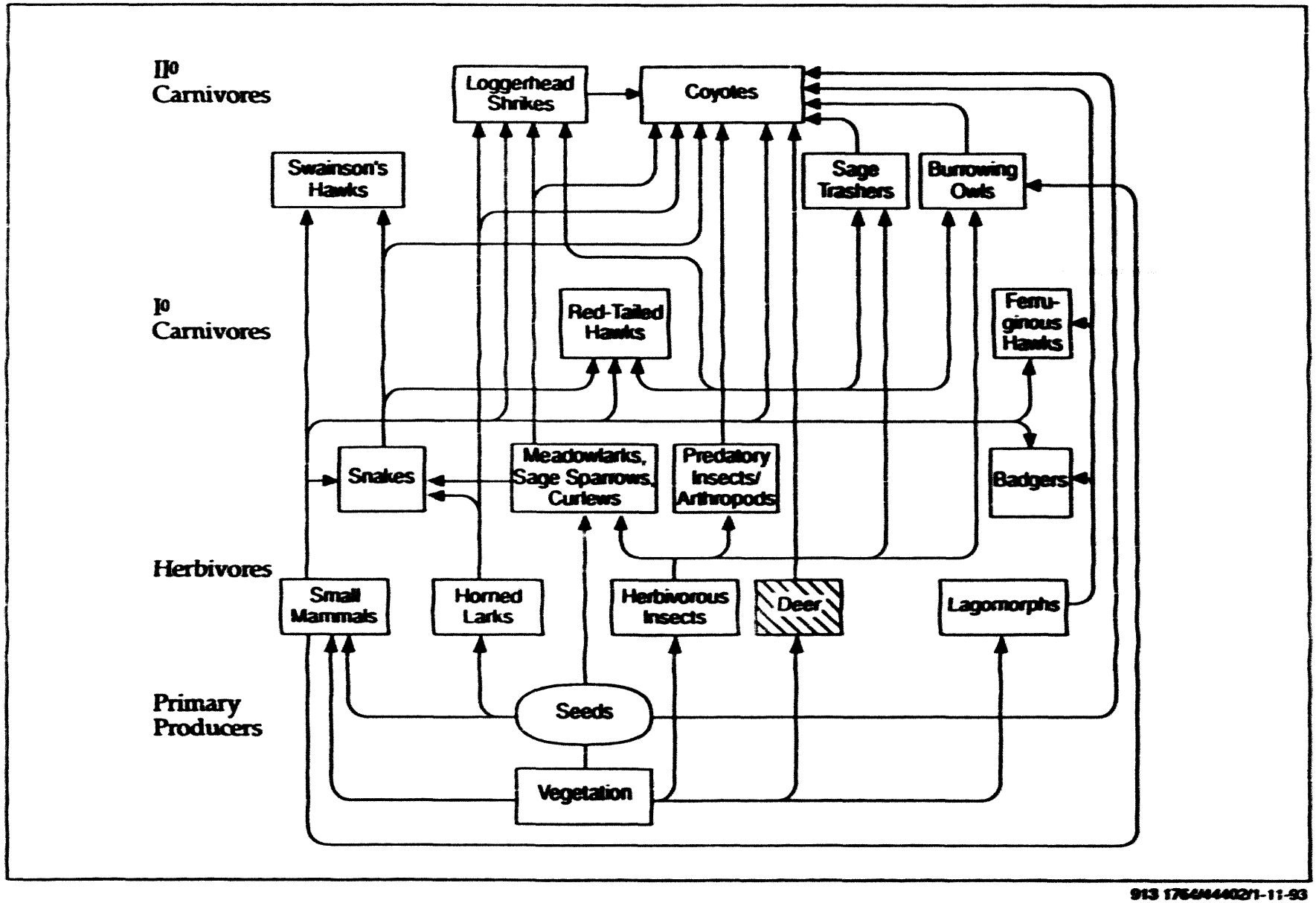

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Figure 6-1. Conceptual Model for the Ecological Risk Assessment of the 200-BP-1 Operable Unit .

Table 6-1. Matrix of Locations, Times, and Exposure Scenarios Evaluated for the 200-BP-1 Operable Unit.

Scenario	On Operable Unit		On Hanford Site (Outside 200 Area)		Off Hanford Site	
	Current	2018	Current	2018	Current	2018
Industrial	Yes	Yes	Yes	Yes	Yes	Yes
Residential	No	No	No	Yes	Yes	Yes
Recreational	No	No	No	Yes	Yes	Yes
Agricultural	No	No	No	Yes	Yes	Yes

Table 6-2. Matrix of Future Exposure Scenarios and Pathways for Contaminants of Potential Concern at the 200-BP-1 Operable Unit.

Media	Pathway[a]	Scenario				
		Industrial on Operable Unit	Industrial on Hanford Site[b]	Residential[b]	Recreational[b]	Agricultural[b]
Soil	Ingestion	yes	no	no	no	no
	Dermal	yes	no	no	no	no
	External	yes	no	no	no	no
Air	Inhalation	yes	yes	yes	yes	yes

[a]None of these pathways are currently complete due to the existence of a clean soil cap. These pathways only become operable in the future scenarios (starting 2018) under the assumption that the soil cap is eroded or otherwise removed.
[b]Direct exposure to contaminated soils by these receptors is not a viable pathway; however, exposure to fugitive dust deposits on the Hanford site (outside of the 200 Area) is assessed.

Table 6-3. Summary of Inhalation Intakes Associated with Measured Air Concentrations of Radionuclides.

Radionuclide	Concentration[a] (pCi/m^3)		Intake[b] (pCi)	
	Adjacent to Operable Unit[c]	Controls Locations[d]	Adjacent to Operable Unit	Control Locations
cesium-137	5.2±2.4E-04	ND	5.2E+01	-
plutonium-238	6.6±6.3E-07[e]	4.2±5.5E-07[e]	6.6E-02	4.2E-02
plutonium-239/240	2.6±1.1E-06	4.5±5.0E-07[e]	2.6E-01	4.5E-02
ruthenium-106	1.7±2.2E-03[e]	ND	1.7E+02	-
strontium-90	2.9±2.1E-05	ND	2.9E+00	-
uranium-234	7.3±2.6E-06	-	7.3E-01	-
uranium-235	3.8±7.2E-07[e]	-	3.8E-02	-
uranium-238	8.0±2.6E-06	-	8.0E-01	-
uranium (total)[f]	-	7.5±3.5E-05	-	7.5E+00

[a] 1991 averages ± 2 times the standard error of the mean.
[b] Intake calculated using industrial exposure parameters and reported air concentrations (ignoring error term).
[c] Source is Schmidt et al. 1992. Data is from air sampler location 967.
[d] Source is Woodruff and Hanf 1992. Data represents a composite from distant communities (Yakima, Sunnyside, and Moses Lake).
[e] Concentration values are less than the two-sigma counting error, indicating that the reported results may have come from a sample with no radioactivity.
[f] Assumed to be naturally-occurring (0.0058% U-234, 0.72% U-235, 99.28% U-238).
ND Indicates not detected.
- Indicates not measured.

Table 6-4. Summary of Industrial Scenario Intakes for Future Exposure to Near Surface Soils of Cribs 216-B-43 through 50 (Receptor on the 200-BP-1 Operable Unit).

Contaminant	Pathway		
	Fugitive Dust Inhalation[a]	Soil Ingestion	External
Radioactive	(pCi)	(pCi)	(pCi-yr/g)
cesium-137	1.3E-02	1.8E+03	2.7E+01
radium-226	2.0E-03	2.9E+02	4.3E+00
strontium-90	9.7E-04	1.4E+02	2.1E+00
thorium-228	6.5E-04	9.5E+01	1.4E+00
uranium	7.7E-04	1.1E+02	1.6E+00

[a] Assumes an airborne particulate concentration of 0.01 $\mu g/m^3$ due to the operable unit.
- Indicates not applicable.

Table 6-5. Summary of Industrial Scenario Intakes for Future Exposure to Near Surface Soils of Crib 216-B-57 (Receptor on the 200-BP-1 Operable Unit).

Contaminant	Pathway		
	Fugitive Dust Inhalation[a]	Soil Ingestion	External
Radioactive	(pCi)	(pCi)	(pCi-yr/g)
cesium-137	2.7E-04	3.9E+01	5.7E-01
radium-226	9.8E-04	1.4E+02	2.1E+00
strontium-90	1.2E-04	1.8E+01	-[b]
thorium-228	7.4E-04	1.1E+02	1.6E+00
uranium	1.4E-03	2.0E+02	3.0E+00

[a]Assumes an airborne particulate concentration of 0.01 $\mu g/m^3$ due to the operable unit.
[b]Not an external exposure hazard.
- Indicates not applicable.

Table 6-6. Summary of Industrial Scenario Intakes for Future Exposure to Infiltration Gravels/Soils of Cribs 216-B-43 through 50 (Receptor on the 200-BP-1 Operable Unit).

Contaminant	Pathway					
	Fugitive Dust Inhalation[a]	Soil Ingestion		Dermal[b]		External[c]
	Carcinogenic	Carcinogenic	Noncarcinogenic	Carcinogenic	Noncarcinogenic	Carcinogenic
Non-Radioactive	-[d]	(mg/kg-d)	(mg/kg-d)	(mg/kg-d)	(mg/kg-d)	-[c]
PCBs	-[d]	1.4E-08	-[e]	1.7E-08	-[e]	-[c]
tributyl phosphate	-[d]	-[d]	4.6E-06	-[d]	9.1E-06	-[c]
cadmium	5.6E-15	-[d]	2.9E-07	-[d]	5.7E-08	-[c]
nickel	-[f]	-[d]	1.2E-05	-[d]	2.4E-06	-[c]
Radioactive	(pCi)	(pCi)	-[b]	-[b]	-[b]	(pCi-yr/g)
antimony-125	5.1E-07	7.5E+00	-[b]	-[b]	-[b]	1.1E-01
cesium-137	1.1E+01	1.6E+08	-[b]	-b	-[b]	2.3E+06
cobalt-60	1.4E-05	2.0E+02	-[b]	-b	-[b]	3.0E+00
plutonium-238	2.6E-04	3.8E+03	-[b]	-b	-[b]	5.5E+01
plutonium-239	2.6E-06	3.8E+01	-[b]	-b	-[b]	5.5E-01
plutonium-239/240	1.1E-02	1.6E+05	-[b]	-b	-[b]	2.3E+03
radium-226	6.9E-05	1.0E+03	-[b]	-b	-[b]	1.5E+01
strontium-90	1.4E+01	2.0E+08	-[b]	-b	-[b]	3.0E+06
technetium-99	8.8E-04	1.3E+04	-[b]	-b	-[b]	1.9E+02
thorium-228	8.5E-06	1.2E+02	-[b]	-b	-[b]	1.8E+00
uranium	6.4E-04	9.3E+03	-[b]	-b	-[b]	1.4E+02

[a] Assumes an airborne particulate concentration of 0.0001 $\mu g/m^3$ due to the operable unit.
[b] Non-radioactive contaminants only.
[c] Radioactive contaminants only.
[d] Slope factor not available to evaluate intake for this pathway.
[e] Reference dose not available to evaluate intake for this pathway.
[f] Nickel is not evaluated for this pathway since only refinery dust is carcinogenic.
- Indicates not applicable.

Table 6-7. Summary of Industrial Scenario Intakes for Future Exposure to Infiltration Gravels/Soils of Crib 216-B-57 (Receptor on the 200-BP-1 Operable Unit).

Contaminant	Pathway					
	Fugitive Dust Inhalation[a]	Soil Ingestion		Dermal[b]		External[c]
	Carcinogenic	Carcinogenic	Non-Carcinogenic	Carcinogenic	Non-Carcinogenic	Carcinogenic
Non-Radioactive	(mg/kg-d)	(mg/kg-d)	(mg/kg-d)	(mg/kg-d)	(mg/kg-d)	-[c]
PCBs	-[d]	3.9E-08	-[e]	4.7E-08	-[e]	-[c]
cadmium	2.7E-15	-[d]	1.4E-07	-[d]	2.8E-08	-[c]
nickel	-[f]	-[d]	2.6E-06	-[d]	5.2E-07	-[c]
Radioactive	(pCi)	(pCi)	-[b]	-[b]	-[b]	(pCi-yr/g)
cesium-137	1.9E-01	2.7E+06	-[b]	-[b]	-[b]	4.0E+04
plutonium-238	9.7E-08	1.4E+00	-[b]	-[b]	-[b]	2.1E-02
plutonium-239	6.3E-08	9.2E-01	-[b]	-[b]	-[b]	1.3E-02
radium-226	1.7E-04	2.5E+03	-[b]	-[b]	-[b]	3.6E+01
strontium-90	2.3E-04	3.3E+03	-[b]	-[b]	-[b]	-[g]
technitium-99	5.0E-04	7.3E+03	-[b]	-[b]	-[b]	1.1E+02
thorium-228	2.1E-05	3.1E+02	-[b]	-[b]	-[b]	4.5E+00
total U	1.1-04	1.6E+03	-[b]	-[b]	-[b]	2.3E+01

[a] Assumes an airborne particulate concentration of 0.0001 $\mu g/m^3$ due- to the operable unit.
[b] Non-radioactive contaminants only.
[c] Radioactive contaminants only.
[d] Slope factor not available to evaluate intake for this pathway.
[e] RfD not available.
[f] Nickel is not evaluated for this pathway since only refinery dust is carcinogenic.
[g] Not an external exposure hazard.
- Indicates not applicable.

Table 6-8. Summary of Industrial Scenario Intakes for Future Exposure to Near Surface Soils of 216 Cribs B-43 through 50 and 216-B-57 (Receptor on the Hanford Site).

Contaminant	Fugitive Dust Inhalation[a]	
	216 B-43 through 50	216-B-57
Radioactive	(pCi)	(pCi)
cesium-137	6.3E-03	1.3E-04
radium-226	1.0E-03	4.9E-04
strontium-90	4.9E-04	6.2E-05
thorium-228	3.3E-04	3.7E-04
uranium	3.9E-04	7.0E-04

[a] The inhalation pathway is the only exposure pathway evaluated for receptors without access to contaminated soils. Intakes are calculated assuming an airborne particulate concentration of 0.005 $\mu g/m^3$ due to the operable unit.

Table 6-9. Summary of Industrial Scenario Intakes for Future Exposure to Infiltration Gravels/Soils of Cribs 216-B-43 through 50 and 216-B-57 (Receptor on the Hanford Site).

Contaminant	Fugitive Dust Inhalation[a]	
	216-B-43 through 50	216-B-57
Non-Radioactive	(mg/kg-d)	(mg/kg-d)
cadmium	2.8E-15	1.4E-15
Radioactive	(pCi)	(pCi)
antimony-125	2.6E-07	-
cesium-137	5.5E+00	9.4E-02
cobalt-60	6.9E-06	-
plutonium-238	1.3E-04	4.9E-08
plutonium-239	1.3E-06	3.2E-08
plutonium-239/240	5.5E-03	-
radium-226	3.5E-05	8.5E-05
strontium-90	7.0E+00	1.1E-04
technetium-99	4.4E-04	2.5E-04
thorium-228	4.3E-06	1.1E-05
uranium	3.2E-04	5.5E-05

[a] The inhalation pathway is the only exposure pathway evaluated for receptors without access to contaminated soils. Intakes are calculated assuming an airborne particulate concentration of 0.00005 $\mu g/m^3$ due to the operable unit.
- Indicates not applicable.

Table 6-10. Summary of Residential and Agricultural Scenario Intakes for Future Exposure to Near Surface Soils of Cribs 216-B-43 through 50 and 216-B-57 (Receptor on the Hanford Site).

Contaminant	Fugitive Dust Inhalation[a]	
	216-B-43 through 50	216-B-57
Radioactive	(pCi)	(pCi)
cesium-137	1.4E-02	3.0E-04
radium-226	2.2E-03	1.1E-03
strontium-90	1.1E-03	1.4E-04
thorium-228	7.1E-04	8.1E-04
uranium	8.4E-04	1.5E-03

[a]The inhalation pathway is the only exposure pathway evaluated for receptors without access to contaminated soils. Intakes are calculated assuming an airborne particulate concentration of 0.005 $\mu g/m^3$ due to the operable unit.

Table 6-11. Summary of Residential and Agricultural Scenario Intakes for Future Exposure to Infiltration Gravels/Soils of Cribs 216-B-43 through 50 and 216-B-57 (Receptor on the Hanford Site).

Contaminant	Fugitive Dust Inhalation[a]	
	216-B-43 through 50	216-B-57
Nonradioactive (carcinogenic)	(mg/kg-d)	(mg/kg-d)
cadmium	6.1E-15	3.0E-15
Radioactive	(pCi)	(pCi)
antimony-125	5.6E-07	-
cesium-137	1.2E+01	2.0E-01
cobalt-60	1.5E-05	-
plutonium-238	2.8E-04	1.1E-07
plutonium-239	2.8E-06	6.9E-08
plutonium-239/240	1.2E-02	-
radium-226	7.6E-05	1.9E-04
strontium-90	1.5E+01	2.5E-04
technetium-99	9.6E-04	5.5E-04
thorium-228	9.3E-06	2.3E-05
uranium	7.0E-04	1.2E-04

[a]The inhalation pathway is the only exposure pathway evaluated for receptors without access to contaminated soils. Intakes are calculated assuming an airborne particulate concentration of 0.00005 $\mu g/m^3$ due to the operable unit.
- Indicates not applicable.

Table 6-12. Summary of Recreational Scenario Intakes for Future Exposure to Near Surface Soils of Cribs 216-B-43 through and 216-B-57 (Receptor on the Hanford Site).

Contaminant	Fugitive Dust Inhalation[a]	
	216-B-43 through 50	216-B-57
Radioactive	(pCi)	(pCi)
cesium-137	2.7E-04	5.7E-06
radium-226	4.2E-05	2.1E-05
strontium-90	2.0E-05	2.6E-06
thorium-228	1.4E-05	1.6E-05
uranium	1.6E-05	2.9E-05

[a]The inhalation pathway is the only exposure pathway evaluated for receptors without access to contaminated soils. Intakes are calculated assuming an airborne particulate concentration of 0.005 $\mu g/m^3$ due to the operable unit.

Table 6-13. Summary of Recreational Scenario Intakes for Future Exposure to Infiltration Gravels/Soils of Cribs 216-B-43 through 50 and 216-B-57 (Receptor on the Hanford Site).

Contaminant	Fugitive Dust Inhalation[a]	
	216-B-43 through 50	216-B-57
Nonradioactive (carcinogenic)	(mg/kg-d)	(mg/kg-d)
cadmium	1.2E-16	5.8E-17
Radioactive	(pCi)	(pCi)
antimony-125	1.1E-08	-
cesium-137	2.3E-01	3.9E-03
cobalt-60	2.9E-07	-
plutonium-238	5.4E-06	2.0E-09
plutonium-239	5.5E-08	1.3E-09
plutonium-239/240	2.3E-04	-
radium-226	1.4E-06	3.6E-06
strontium-90	2.9E-01	4.8E-06
technetium-99	1.8E-05	1.1E-05
thorium-228	1.8E-07	4.4E-07
uranium	1.3E-05	2.3E-06

[a]The inhalation pathway is the only exposure pathway evaluated for receptors without access to contaminated soils. Intakes are calculated assuming an airborne particulate concentration of 0.00005 $\mu g/m^3$ due to the operable unit.
- Indicates not applicable.

Table 6-14. Summary of Systemic Toxicity Information for Contaminants of Potential Concern at the 200-BP-1 Operable Unit.

Contaminant	Oral RfD mg/kg-d	Oral RfD basis/source	Confidence Level	Critical Effect	Uncertainty Factors	Modifying Factors
cadmium	1.0E-03	food/IRIS[a]	high	proteinuria	10	1
nickel	2.0E-02	diet/IRIS[a]	medium	decreased body and organ weights	300	1
PCBs	ND	-	-	-	-	-
tributyl phosphate	5E-03	food/STSC[b]	low	urinary bladder hyperplasia	3,0000	1

Note: Inhalation RfDs have not been developed for any of these contaminants.
[a]Integrated Risk Information System (IRIS; EPA 1993)
[b]Superfund Technical Support Center (STSC; EPA 1992c)
RfD = Reference Dose
ND Indicates not determined
- Indicates not applicable

Table 6-15. Summary of Carcinogenic Toxicity Information for Contaminants of Potential Concern at the 200-BP-1 Operable Unit.

Contaminant	Weight of Evidence	Type of Cancer	Oral SF	Inhalation SF	External SF
Nonradioactive			$(mg/kg\text{-}d)^{-1}$	$(mg/kg\text{-}d)^{-1}$	NA
cadmium	B1	lung, trachea bronchial tumors	-	6.1E+00[a]	NA
PCBs	B2	liver	7.7[a]	-	NA
Radioactive			$(pCi)^{-1}$	$(pCi)^{-1}$	$(pCi\text{-}yr/g)^{-1}$
antimony-125	A	-	8.4E-13[b]	1.1E-11[b]	1.2E-06[b]
cesium-137	A	-	2.8E-11[b]	1.9E-11[b]	2.0E-06[b]
cobalt-60	A	-	1.5E-11[b]	1.5E-10[b]	8.6E-06[b]
plutonium-238	A	-	2.2E-10[b]	3.9E-08[b]	2.8E-11[b]
plutonium-239	A	-	2.3E-10[b]	3.8E-08[b]	1.7E-11[b]
radium-226	A	bone	1.2E-10[b]	3.0E-09[b]	6.0E-06[b]
strontium-90	A	-	3.6E-11[b]	6.2E-11[b]	0.0E+00[b,c]
technetium-99	A	-	1.3E-12[b]	8.3E-12[b]	6.0E-13[b]
thorium-228	A	-	5.5E-11[b]	7.8E-08[b]	5.6E-06[b]
uranium-238	A	-	2.8E-11[b]	5.2E-08[b]	3.6E-08[b]

[a]PCB congeners vary greatly as to their potency - Aroclor is assumed to be representative of all PCB mixtures (EPA 1993).
[b]Health Effects Assessment Summary Tables (HEAST; EPA 1992b). Radionuclides slope factors include contribution from daughter products.
[c]Sr-90 is not considered an external hazard.
SF = Slope Factor.
- Indicates not available.
NA Indicates not applicable.

Table 6-16. Dermally Adjusted Slope Factors (SF) and Reference Doses (RfD) for Contaminants of Potential Concern at the 200-BP-1 Operable Unit.

Contaminant	Oral Absorption Factor	Adjusted Dermal SF $(mg/kg\text{-}d)^{-1}$	Adjusted Dermal RfD (mg/kg-d)	Reference
cadmium	0.05	-[a]	5.0E-05	EPA 1992b
nickel	0.05	-[a]	1.0E-03	EPA 1992b
PCBs	0.90	8.6E+00	-[a]	SRC 1991b
tributyl phosphate	ND	-[a]	5.0E-03[b]	-

[a] Oral RfD or SF not published by EPA.
[b] Oral RfD is used as the dermal RfD without adjustment.
ND Indicates not determined.
- Indicates not applicable.

Table 6-17. Summary of the Baseline Industrial Scenario Risk Assessment for Future Exposure to Near Surface Soils of Cribs 216-B-43 through 50 (Receptor on the 200-BP-1 Operable Unit).

Contaminant	Pathway			Contaminant Total	Exposure Setting Total
	Fugitive Dust Inhalation	Soil Ingestion	External[b]		
	ICR	ICR	ICR	ICR	ICR
Radioactive					
cesium-137	2E-13	5E-08	5E-05	5E-05	
radium-226	6E-12	4E-08	3E-05	3E-05	
strontium-90	6E-14	5E-09	-[a]	5E-09	
thorium-228	5E-11	5E-09	8E-06	8E-06	
uranium	4E-11	3E-09	6E-08	6E-08	
Pathway Total	1E-10	1E-07	9E-05		9E-05

ICR = Lifetime Incremental Cancer Risk
[a] Not an external exposure hazard
- Indicates not applicable.

Table 6-18. Summary of the Baseline Industrial Scenario Risk Assessment for Future Exposure to Near Surface Soils of Crib 216-B-57 (Receptor on the 200-BP-1 Operable Unit).

Contaminant	Pathway			Contaminant Total	Exposure Setting Total
	Fugitive Dust Inhalation	Soil Ingestion	External		
	ICR	ICR	ICR	ICR	ICR
Radioactive					
cesium-137	5E-15	1E-09	1E-06	1E-06	
radium-226	3E-12	2E-08	1E-05	1E-05	
strontium-90	8E-15	7E-10	-[a]	7E-10	
thorium-228	6E-11	6E-09	9E-06	9E-06	
uranium	7E-11	6E-09	1E-07	1E-07	
Pathway Total	1E-10	3E-08	2E-05		2E-05

ICR = Lifetime Incremental Cancer Risk
[a]Not an external exposure hazard
- Indicates not applicable.

Table 6-19. Summary of the Baseline Industrial Scenario Risk Assessment for Future Exposure to Infiltration Gravels/Soils of Cribs 216-B-43 through 50 (Receptor on the 200-BP-1 Operable Unit).

Contaminant	Pathway						Contaminant Total		Exposure Setting Total	
	Fugitive Dust Inhalation	Soil Ingestion		Dermal[a]		External[b]				
	ICR	ICR	HQ	ICR	HQ	ICR	ICR	HQ	ICR	HQ
Non-Radioactive										
PCBs	-[c]	1E-07	-[d]	1E-07	-[d]	-[b]	2E-07	-[d]		
tributyl phosphate	-[c]	-[c]	0.0009	-[c]	0.002	-[b]	-[c]	0.003		
cadmium	4E-14	-[c]	0.0003	-[c]	0.001	-[b]	4E-14	0.001		
nickel	-[g]	-[c]	0.0006	-[c]	0.002	-[b]	-	0.003		
Radioactive										
antimony-125	6E-18	6E-12	-[a]	-[a]	-[a]	1E-07	1E-07	-[a]		
cesium-137	2E-10	5E-03	-[a]	-[a]	-[a]	>1E-02[e]	>1E-02[e]	-[a]		
cobalt-60	2E-15	3E-09	-[a]	-[a]	-[a]	3E-05	3E-05	-[a]		
plutonium-238	1E-11	8E-07	-[a]	-[a]	-[a]	2E-09	8E-07	-[a]		
plutonium-239	1E-13	9E-09	-[a]	-[a]	-[a]	9E-12	9E-09	-[a]		
plutonium-239/240	4E-10	4E-05	-[a]	-[a]	-[a]	4E-08	4E-05	-[a]		
radium-226	2E-13	1E-07	-[a]	-[a]	-[a]	9E-05	9E-05	-[a]		
strontium-90	9E-10	7E-03	-[a]	-[a]	-[a]	-[f]	7E-03	-[a]		
technetium-99	7E-15	2E-08	-[a]	-[a]	-[a]	1E-10	2E-08	-[a]		
thorium-228	7E-13	7E-09	-[a]	-[a]	-[a]	1E-05	1E-05	-[a]		
uranium	3E-11	3E-07	-[a]	-[a]	-[a]	5E-06	5E-06	-[a]		
Pathway Total	2E-09	1E-02	0.002	1E-07	0.005	>1E-02[e]			>1E-02[e]	0.007

[a] Non-radioactive contaminants only.
[b] Radioactive contaminants only.
[c] SF not available to evaluate this pathway.
[d] RfD not available to evaluate this pathway.
[e] ICR exceeds the range within which the assumption of a linear dose-response is valid.
[f] Not an external exposure hazard.
[g] Nickel is not evaluated for this pathway since only refinery dust is carcinogenic.
ICR = Lifetime Incremental Cancer Risk
HQ = Hazard Quotient
- Indicates not applicable.

Table 6-20. Summary of the Baseline Industrial Scenario Risk Assessment for Future Exposure to Infiltration Gravels/Soils of Crib 216-B-57 (Receptor on the 200-BP-1 Operable Unit).

Contaminant	Pathway						Contaminant Total		Exposure Setting Total	
	Fugitive Dust Inhalation	Soil Ingestion		Dermal[a]		External[b]				
	ICR	ICR	HQ	ICR	HQ	ICR	ICR	HQ	ICR	HQ
Non-Radioactive										
PCBs	-[c]	3E-07	-[d]	4E-07	-[d]	-[b]	7E-07	-[d]		
cadmium	2E-14	-[c]	0.0001	-[c]	0.0006	-[b]	2E-14	0.0007		
nickel	-[g]	-[c]	0.0001	-[c]	0.0005	-[b]	-[b]	0.0006		
Radioactive										
cesium-137	4E-12	8E-05	-[a]	-[a]	-[a]	>1E-02[e]	>1E-02[e]	-[a]		
plutonium-238	4E-15	3E-10	-[a]	-[a]	-[a]	6E-13	3E-10	-[a]		
plutonium-239	2E-15	2E-10	-[a]	-[a]	-[a]	2E-13	2E-10	-[a]		
radium-226	5E-13	3E-07	-[a]	-[a]	-[a]	2E-04	2E-04	-[a]		
strontium-90	1E-14	1E-07	-[a]	-[a]	-[a]	-[f]	1E-07	-[a]		
technetium-99	4E-15	1E-08	-[a]	-[a]	-[a]	6E-11	1E-08	-[a]		
thorium-228	2E-12	2E-08	-[a]	-[a]	-[a]	3E-05	3E-05	-[a]		
uranium	6E-12	5E-08	-[a]	-[a]	-[a]	8E-07	8E-07	-[a]		
Pathway Total	1E-11	8E-05	0.0002	4E-07	0.001	>1E-02[e]			>1E-02[e]	0.001

[a] Non-radioactive contaminants only.
[b] Radioactive contaminants only.
[c] SF not available to evaluate this pathway.
[d] RfD not available to evaluate this pathway.
[e] ICR exceeds the range within which the assumption of a linear dose-response is valid.
[f] Not an external exposure hazard.
[g] Nickel is not evaluated for this pathway since only refinery dust is carcinogenic.
ICR = Lifetime Incremental Cancer Risk.
HQ = Hazard Quotient
- Indicates not applicable.

Table 6-21. Summary of Inhalation Cancer Risks Associated with Measured Air Concentrations of Radionuclides.

Radionuclide	Risk[a]	
	Adjacent to Operable Unit[b]	Control Locations[c]
cesium-137	1E-09	-
plutonium-238	3E-09[d]	2E-09[d]
plutonium-239/240	1E-08	2E-09[d]
ruthenium-106[e]	8E-08[d]	-
strontium-90	2E-10	-
uranium-234[e]	2E-08	-
uranium-235[e]	1E-09[d]	-
uranium-238	4E-08	-
uranium (total)[e]	-	3E-07
Total Risk	2E-07	3E-07

[a]Lifetime cancer risk using industrial exposure parameters and reported air concentrations (ignoring error term).
[b]Air sampler location 967 (Schmidt et al. 1992).
[c]Composite from distant communities (Yakima, Sunnyside, and Moses Lake; Woodruff and Hanf 1992).
[d]Based on concentration values that are less than the two-sigma counting error, indicating that the reported results may have come from a sample with no radioactivity.
[e]Inhalation SFs for Ru-106, U-234, U-235, and U(total) are 4.4E-10, 2.6E-08, 2.5E-08 and 3.8E-08 (pCi)-1, respectively (EPA 1992b).
- Indicates radionuclide not measured.

Table 6-22. Summary of Total Cancer Risks Associated with Current Conditions at the 200-BP-1 Operable Unit.

Location	Pathway			Total
	Inhalation	External		
		Control	Cribs	
200-BP-1 Operable Unit				
Cribs 216-B-43 through 50	2E-07[a]	2E-04[a]	1E-07[b]	2E-04
Crib 216-B-57	2E-07[a]	2E-04[a]	6E-08[b]	2E-04
Control Locations[c]	3E-07[a]	2E-04[a]	-	2E-04

[a]Attribute to sources other than the operable unit, see Section 6.2.3.2.1.
[b]Attributable to operable unit contaminants.
[c]Composite from distance communities (Yakima, Sunnyside, and Moses Lake).
Note: Inhalation and external control risks are based on environmental monitoring data. External control risks are largely attributable to naturally occurring radionuclides in soil, and are not due to Hanford Site contaminants. See Section 6.2.3.2.1 for additional information.

Table 6-23. Summary of the Baseline Industrial Scenario Risk Assessment for Future Exposure to Near Surface Soils of Cribs 216-B-43 Through 50 and 216-B-57 (Receptor on the Hanford Site).

Contaminant	Fugitive Dust Inhalation[a]		Contaminant Total	Exposure Setting Total
	216-B-43 through 50	216-B-57		
	ICR	ICR	ICR	ICR
Radioactive				
cesium-137	1E-13	3E-15	-	
radium-226	3E-12	2E-12	-	
strontium-90	3E-14	4E-15	-	
thorium-228	3E-11	3E-11	-	
uranium	2E-11	4E-11	-	
Pathway Total	5E-11	7E-11		1E-10

[a]The inhalation pathway is the only exposure pathway evaluated for receptors without access to contaminated soils.
ICR = Lifetime Incremental Cancer Risk

Table 6-24. Summary of the Baseline Industrial Scenario Risk Assessment for Future Exposure to Infiltration Gravels/Soils of Cribs 216-B-43 through 50 and 216-B-57 (Receptor on the Hanford Site).

Contaminant	Fugitive Dust Inhalation[a]		Contaminant Total	Exposure Setting Total
	216-B-43 through 50	216-B-57		
	ICR	ICR	ICR	ICR
Non-Radioactive				
cadmium	2E-14	9E-15	3E-14	
Radioactive				
antimony-125	3E-18	-	3E-18	
cesium-137	1E-10	2E-12	1E-10	
cobalt-60	1E-15	-	1E-15	
plutonium-238	5E-12	2E-15	5E-12	
plutonium-239	5E-14	1E-15	5E-14	
plutonium-239/240	2E-10	-	2E-10	
radium-226	1E-13	3E-13	4E-13	
strontium-90	4E-10	7E-15	4E-10	
technetium-99	4E-15	2E-15	6E-15	
thorium-228	3E-13	8E-13	1E-12	
uranium	2E-11	3E-12	2E-11	
Pathway Total	7E-10	6E-12		7E-10

[a]The inhalation pathway is the only exposure pathway evaluated for receptors without access to contaminated soils.
ICR = Lifetime Incremental Cancer Risk
- Indicates not applicable.

Table 6-25. Summary of the Baseline Residential and Agricultural Scenario Risk Assessments for Future Exposure to Near Surface Soils of Cribs 216-B-43 through 50 and 216-B-57 (Receptor on the Hanford Site).

Contaminant	Fugitive Dust Inhalation[a]		Contaminant Total	Exposure Setting Total
	216-B-43 through 50	216-B-57		
	ICR	ICR	ICR	ICR
Radioactive				
cesium-137	3E-13	6E-15	3E-13	
radium-226	7E-12	3E-12	1E-11	
strontium-90	7E-14	8E-15	8E-14	
thorium-228	6E-11	6E-11	1E-10	
uranium	4E-11	8E-11	1E-10	
Pathway Total	1E-10	1E-10		2E-10

[a]The inhalation pathway is the only exposure pathway evaluated for receptors without access to contaminated soils.
ICR = Lifetime Incremental Cancer Risk

Table 6-26. Summary of the Baseline Residential and Agricultural Scenario Risk Assessments for Future Exposure to Infiltration Gravels/Soils of Cribs 216-B-43 through 50 and 216-B-57 (Receptor on the Hanford Site).

Contaminant	Fugitive Dust Inhalation[a]		Contaminant Total	Exposure Setting Total
	216-B-43 through 50	216-B-57		
	ICR	ICR	ICR	ICR
Nonradioactive				
cadmium	4E-14	2E-14	6E-14	
Radioactive				
antimony-125	6E-18	-	6E-18	
cesium-137	2E-10	4E-12	2E-10	
cobalt-60	2E-15	-	2E-15	
plutonium-238	1E-11	4E-15	1E-11	
plutonium-239	1E-13	3E-15	1E-13	
plutonium-239/240	5E-10	-	5E-10	
radium-226	2E-13	6E-13	8E-13	
strontium-90	1E-09	2E-14	1E-09	
technetium-99	8E-15	5E-15	1E-14	
thorium-228	7E-13	2E-12	3E-12	
uranium	4E-11	6E-12	4E-11	
Pathway Total	2E-09	1E-11		2E-09

[a]The inhalation pathway is the only exposure pathway evaluated for receptors without access to contaminated soils.
ICR = Lifetime Incremental Cancer Risk
- Indicates not applicable.

Table 6-27. Summary of the Baseline Recreational Scenario Risk Assessment for Future Exposure to Near Surface Soils of Cribs 216-B-43 through 50 and 216-B-57 (Receptor on the Hanford Site).

Contaminant	Fugitive Dust Inhalation[a]		Contaminant Total	Exposure Setting Total
	216-B-43 through 50	216-B-57		
	ICR	ICR	ICR	ICR
Radioactive				
cesium-137	5E-15	1E-16	5E-15	
radium-226	1E-13	6E-14	1E-13	
strontium-90	1E-15	2E-16	1E-15	
thorium-228	1E-12	1E-12	2E-12	
uranium	8E-13	2E-12	3E-12	
Pathway Total	2E-12	3E-12		5E-12

[a]The inhalation pathway is the only exposure pathway evaluated for receptors without access to contaminated soils.
ICR = Lifetime Incremental Cancer Risk

Table 6-28. Summary of the Baseline Recreational Scenario Risk Assessment for Future Exposure to Infiltration Gravels/Soils of Cribs 216-B-43 through 50 and 216-B-57 (Receptor on the Hanford Site).

Contaminant	Fugitive Dust Inhalation[a]		Contaminant Total	Exposure Setting Total
	216-B-43 through 50	216-B-57		
	ICR	ICR	ICR	ICR
Nonradioactive				
cadmium	7E-16	4E-16	1E-15	
Radioactive				
antimony-125	1E-19	-	1E-19	
cesium-137	4E-12	8E-14	4E-12	
cobalt-60	4E-17	-	4E-17	
plutonium-238	2E-13	8E-17	2E-13	
plutonium-239	2E-15	5E-17	2E-15	
plutonium-239/240	9E-12	-	9E-12	
radium-226	4E-15	1E-14	1E-14	
strontium-90	2E-11	3E-16	2E-11	
technetium-99	2E-16	9E-17	3E-16	
thorium-228	1E-14	3E-14	4E-14	
uranium	7E-13	1E-13	8E-13	
Pathway Total	3E-11	2E-13		3E-11

[a]The inhalation pathway is the only exposure pathway evaluated for receptors without access to contaminated soils.

ICR = Lifetime Incremental Cancer Risk
- Indicates not applicable.

Table 6-29. Exposure Pathways and Contaminants Resulting from Fugitive Dust Deposition Evaluated for Future Receptors on the Hanford Site.

Media	Pathway	Contaminants
Soil	Ingestion External	cesium-137, strontium-90 cesium-137
Biota	Plant/Crop Dairy Beef	cesium-137, strontium-90 cesium-137, strontium-90 cesium-137, strontium-90

Table 6-30. Risk Assessment Summary of Exposure Pathways Resulting from Fugitive Dust Deposition for Future Receptors on the Hanford Site.

Contaminant	Pathway					Contaminant Total
	Soil Ingestion	Plant	Meat	Milk	External	
Infiltration Gravels/Soils Cribs 216-B-43 to 50						
cesium-137	3E-11	5E-12	1E-11	7E-12	1E-07	1E-07
strontium-90	9E-11	2E-09	4E-11	7E-10	-	2E-09
Infiltration Gravels/Soils Crib 216-B-57						
cesium-137	5E-13	9E-14	2E-13	1E-13	2E-09	2E-09
strontium-90	2E-15	3E-14	8E-16	1E-14	-	4E-14

Note: All values represent lifetime incremental cancer risks calculated by using agricultural scenario exposure parameters, and assuming a contamination zone thickness of 0.02 mm.
- Indicates not applicable.

Table 6-31. Risk Assessment Summary Table of Incremental Cancer Risk Estimates for Non-Radioactive and Radioactive Contaminants under Current Conditions (by Crib Grouping).

Scenario	On Operable Unit	On Hanford Site	Off Hanford Site
Cribs 216-B-43 to 50			
Industrial	1E-07	NA	-
Residential	-	-	NA
Recreational	-	-	NA
Agricultural	-	-	NA
Crib 216-B-57			
Industrial	6E-08	NA	-
Residential	-	-	NA
Recreational	-	-	NA
Agricultural	-	-	NA

NA indicates not applicable. External exposure is the only pathway, and does not impact receptors at these locations.
- Indicates not evaluated.

Table 6-32. Risk Assessment Summary Table of Incremental Cancer Risk Estimates for Non-Radioactive and Radioactive Contaminants for Future Settings (by Crib Grouping).

Scenario	On Operable Unit			On Hanford Site			Off Hanford Site		
	Cover In Place	Near Surface Soils Exposed	Infiltration Gravels/Soils Exposed	Cover In Place	Near Surface Soils Exposed	Infiltration Gravels/Soils Exposed[a]	Cover In Place	Near Surface Soils Exposed	Infiltration Gravels/Soils Exposed
Cribs 216-B-43 through 50									
Industrial	4E-07	9E-05	>1E-02	NA	5E-11	1E-08	-	-	-
Residential	-	-	-	-	1E-10	1E-07	NA	-[b]	-[b]
Recreational	-	-	-	-	2E-12	6E-10	NA	-[b]	-[b]
Agricultural	-	-	-	-	1E-10	1E-07	NA	-[b]	-[b]
Crib 216-B-57									
Industrial	1E-07	2E-05	>1E-02	NA	7E-11	2E-10	-	-	-
Residential	-	-	-	-	1E-10	2E-09	NA	-[b]	-[b]
Recreational	-	-	-	-	3E-12	1E-11	NA	-[b]	-[b]
Agricultural	-	-	-	-	1E-10	2E-09	NA	-[b]	-[b]

[a] Cancer risk estimates include contribution from fugitive dust deposition.
[b] These (inhalation) risks were not calculated because of the extreme dilution that would occur before contaminants would reach off-site locations.
NA Indicates not applicable. External exposure is the only operative pathway, and does not impact receptors at these locations.
- Indicates not evaluated.

Table 6-33. Exposure Pathways Evaluated for Terrestrial Wildlife Indicator Species.

Indicator Species	Pathways Evaluated
Pocket Mouse	Plant ingestion
Jack Rabbit	Plant ingestion
Coyote	Plant ingestion Small mammal ingestion
Burrowing Owl	Small mammal ingestion Insect ingestion
Loggerhead Shrike	Small mammal ingestion Insect ingestion
Swainson's Hawk	Small mammal ingestion Insect ingestion

Table 6-34. Wildlife Exposure Parameters Used in the 200-BP-1 Ecological Risk Assessment.

Species	Weight (kg)	Ingestion rate (kg/day) wet weight	Effective Radius (cm)[8]
Great Basin Pocket Mouse	.021	1.327[3]	2
Jack Rabbit	2.5[2]	1.18[4]	7
Swainson's Hawk	1.05[6]	.46[7]	7
Loggerhead Shrike	.04[6]	.048[7]	3
Burrowing owl	.150[6]	.112[7]	3
Coyote	12[2]	2.02[8]	20

(1) Burt and Grossenheider 1976.
(2) Chapman and Feldhamer 1982.
(3) FEMP, Site-wide Characterization Report, 1992 for white-footed mouse of the same weight.
(4) Scaled from white-footed mouse weight and ingestion rate from FEMP, Site-wide Characterization Report, 1992, using the following formula: metabolic rate = alpha x (body weight) $^{0.75}$ (Kleiber 1961).
(5) Wallmo 1981.
(6) Terres 1980.
(7) Scaled from red-tailed hawk weight and ingestion rate from FEMP, Site-wide Characterization Report 1992, using the following formula: metabolic rate = alpha x (body weight) $^{0.73}$ (Calder 1984).
(8) Scaled from red fox weight and ingestion rate from FEMP, Site-wide Characterization Report, 1992, using the following formula: metabolic rate = alpha x (body weight) $^{0.75}$ (Kleiber 1961).
(9) To determine the effective radius of an animal we calculated the radius of a spherical animal using the weight of the animal and assuming a density of animal of 1 g/cm^3 [9](see table below). We used the equation for the volume of a sphere: volume = 4/3 π r^3.

For References 4, 7, and 8 above we determined the alpha constant by substituting known weights and ingestion rates into the formula, then using that alpha to determine the ingestion rate for the animal of known size. For example, to calculate the ingestion rate (metabolic rate) of the jack rabbit, we used the white-footed mouse weight and ingestion rate. 0.0327= alpha * (0.021)0.75 alpha = .5928. ingestion rate for the rabbit therefore = .5928 * $(2.5)^{0.75}$ = 1.18.

Table 6-35. Soil-to-Plant and Soil-to-Insect Transfer Coefficients Used for Constituents of Potential Concern.

Contaminant	Transfer Coefficient	Reference
Soil to Plant:		
Cs-137	0.62	Miller et al. 1977
Ra-226	0.1	Coughtrey et al. 1983 and 1985
Th-228	1.00E-4	Whicker and Schultz 1982
Total U	1	Miller et al. 1977
Sr-90/Y-90	19	Rouston and Cataldo 1978
Soil to Insect:		
All Contaminants Listed Above	1	assumption

Table 6-36. Dry-Weight to Wet-Weight Conversion Factors.

Item	Conversion Factor	Reference
Vegetation	0.4	Brandt and Rickard, 1992
Insects	0.3	Brandt, personal communication, 1992

Table 6-37. Literature Review of Transfer Coefficients for Constituent of Concern in the 200-BP-1 Operable Unit.

Element	Plant/Soil Transfer radio	Animal/plant, soil transfer ratio	Carnivore/herbivore transfer ratio
Cesium-137	native veg (Hanford) = 0.62° Russian thistle = 0.025-0.053[1] 0.25[2]	$GI_{monogastric}$ = 1.0[2] $GI_{ruminants}$ = 0.6[2] Great Basin pocket mouse = 0.13+	Cougar/deer = 3.4[4] .003*
Radium-226	0.01-0.1#	0.01[3]	7.9E-4*
Thorium-228	$<10^{-4}$#	Coef of transfer $_{(poultry)}$ = 4.0 E-3ˆ	
Total U	10^{-1}-10^{-4}# 0.1°	deer/plant = 3.8 E-4	
Strontium-90	Russian thistle = 10.8 - 19[1] 0.017# 5.23°	GI = 0.05-0.4 (range)[2] GI = 0.2 (best estimate)[2] up to 1.0 in preweanlings[2]	2.04E-4*

[1]Routson and Cataldo 1978.
[2]Coughtrey et al. 1983.
[3]Reichle et al. 1970.
[4]Pendelton et al. 1964.
[5]Kabata-Pendias and Pendias, 1984.
+ Health Physics Symposium 1979.
o Miller et al. 1977.
US NRC Reg (Whicker and Schultz 1982).
* Calculated from muscle (fox)/whole body (mice) from (DOE, 1992), and values found in Coughtrey et al. 1983.
ˆSoldat, 1992
GI = Fractional Gastrointestinal Absorption.

Table 6-38. Estimated Doses for Great Basin Pocket Mice.

Contaminant	Activity/g soil (pCi/g)	Activity/kg vegetation (wet) (Ci/kg)	Dose rate (rad/day)	Fractional use	Dose rate (rad/day)
Cs-137	23	5.704E-09	1.28E-03	1	1.28E-03
Ra-226	2.08	8.32E-11	5.24E-03	1	5.24E-03
Th-228	0.74	2.96E-14	8.07E-10	1	8.07E-10
Total U	1.4	5.6E-10	1.36E-03	1	1.36E-03
Sr-90/Y-90	1.8	1.064E-08	1.51E-01	1	1.51E-01
Total			1.59E-01		1.59E-01

Table 6-39. Estimated Doses for Jack Rabbits.

Contaminant	Activity/g soil (pCi/g)	Activity/kg vegetation (wet) (Ci/kg)	Dose rate (rad/day)	Fractional use	Dose rate (rad/day)
Cs-137	23	5.704E-09	1.58E-03	1	1.58E-03
Ra-226	2.08	8.32E-11	1.59E-03	1	1.59E-03
Th-228	0.74	2.96E-14	2.45E-10	1	2.45E-10
Total U	1.4	5.6E-10	4.13E-04	1	4.13E-04
Sr-90/Y-90	1.8	1.064E-08	6.42E-02	1	6.42E-02
Total			6.78E-02		6.78E-02

Table 6-40. Estimated Doses for Swainson's Hawks.

Ingestion of mice

Contaminant	Activity/g soil (pCi/g)	Activity/kg vegetation (wet) (Ci/kg)	Activity/kg mouse (Ci/kg)	Dose rate (rad/day)	Fractional use	Dose rate (rad/day)
Cs-137	23	5.704E-09	9.36E-08	1.95E-02	0.018	3.50E-04
Ra-226	2.08	8.32E-11	9.30E-09	1.65E-01	0.018	2.97E-03
Th-228	0.74	2.96E-14	2.81E-15	2.16E-11	0.018	3.89E-13
Total U	1.4	5.6E-10	5.77E-09	3.96E-03	0.018	7.12E-05
Sr-90/Y-90	1.8	1.064E-08	1.10E-06	5.93E+00	0.018	1.07E-01
Total				6.12E+00		1.10E-01

Ingestion of insects

Contaminant	Activity/g soil (pCi/g)	Activity/kg insects (wet) (Ci/kg)	Dose rate (rad/day)	Fractional use	Dose rate (rad/day)
Cs-137	23	6.90E-09	1.43E-03	0.018	2.58E-05
Ra-226	2.08	6.24E-10	1.11E-02	0.018	1.99E-04
Th-228	0.74	2.22E-10	1.70E-06	0.018	3.07E-08
Total U	1.4	4.20E-10	2.88E-04	0.018	5.18E-06
Sr-90/Y-90	1.8	4.20E-10	2.32E-03	0.018	4.18E-05
Total			2.01E-02		2.72E-04

Table 6-41. Estimated Doses for Loggerhead Shrikes.

Ingestion of mice.

Contaminant	Activity/g soil (pCi/g)	Activity/kg vegetation (wet) (Ci/kg)	Activity/kg mouse (Ci/kg)	Dose rate (rad/day)	Fractional use	Dose rate (rad/day)
Cs-137	23	5.704E-09	9.36E-08	1.76E-02	0.35	6.17E-03
Ra-226	2.08	8.32E-11	9.30E-09	3.85E-01	0.35	1.35E-01
Th-228	0.74	2.96E-14	2.81E-15	5.04E-11	0.35	1.76E-11
Total U	1.4	5.6E-10	5.77E-09	1.69E-02	0.35	5.93E-03
Sr-90/Y-90	1.8	1.064E-08	1.10E-06	1.62E+01	0.35	5.67E+00
Total				1.66E+01		5.80E+00

Ingestion of insects.

Contaminant	Activity/g soil (pCi/g)	Activity/kg insects (wet) (Ci/kg)	Dose rate (rad/day)	Fractional use	Dose rate (rad/day)
Cs-137	23	6.90E-09	1.30E-03	0.35	4.55E-04
Ra-226	2.08	6.24E-10	2.58E-02	0.35	9.03E-03
Th-228	0.74	2.22E-10	3.97E-06	0.35	1.39E-06
Total U	1.4	4.20E-10	1.23E-03	0.35	4.31E-04
Sr-90/Y-90	1.8	4.20E-10	6.48E-03	0.35	2.23E-03
Total			3.48E-02		1.22E-02

Table 6-42. Estimated Doses for Burrowing Owls.

Ingestion of mice.

Contaminant	Activity/g soil (pCi/g)	Activity/kg vegetation (wet) (Ci/kg)	Activity/kg mouse (Ci/kg)	Dose rate (rad/day)	Fractional use	Dose rate (rad/day)
Cs-137	23	5.704E-09	9.36E-08	1.76E-02	0.26	4.57E-03
Ra-226	2.08	8.32E-11	9.30E-09	2.81E-01	0.26	7.31E-02
Th-228	0.74	2.96E-14	2.81E-15	3.68E-11	0.26	9.57E-12
Total U	1.4	5.6E-10	5.77E-09	1.37E-02	0.26	3.56E-03
Sr-90/Y-90	1.8	1.064E-08	1.10E-06	11.0E+00	0.26	2.86E+00
Total				11.3E+00		2.94E+00

Ingestion of insects.

Contaminant	Activity/g soil (pCi/g)	Activity/kg insects (wet) (Ci/kg)	Dose rate (rad/day)	Fractional use	Dose rate (rad/day)
Cs-137	23	6.90E-09	1.30E-03	0.26	3.37E-04
Ra-226	2.08	6.24E-10	1.89E-02	0.26	4.90E-03
Th-228	0.74	2.22E-10	2.90E-06	0.26	7.55E-07
Total U	1.4	4.20E-10	9.95E-04	0.26	2.59E-04
Sr-90/Y-90	1.8	4.20E-10	4.0E-03	0.26	1.04E-03
Total			2.86E-02		6.54E-03

Table 6-43. Estimated Doses for Coyotes.

Ingestion of vegetation.

Contaminant	Activity/g soil (pCi/g)	Activity/kg vegetation (wet) (Ci/kg)	Dose rate (rad/day)	Fractional use	Dose rate (rad/day)
Cs-137	23	5.704E-09	1.15E-03	0.04	4.62E-05
Ra-226	2.08	8.32E-11	5.61E-04	0.04	2.24E-05
Th-228	0.74	2.96E-14	8.64E-11	0.04	3.46E-12
Total U	1.4	5.6E-10	1.46E-04	0.04	5.84E-06
Sr-90/Y-90	1.8	1.064E-08	2.39E-02	0.04	9.56E-04
Total			2.57E-02		1.03E-03

Ingestion of mice.

Contaminant	Activity/g soil (pCi/g)	Activity/kg vegetation (wet) (Ci/kg)	Activity/kg mouse (Ci/kg)	Dose rate (rad/day)	Fractional use	Dose rate (rad/day)
Cs-137	23	5.704E-09	9.36E-08	1.90E-02	0.04	7.58E-04
Ra-226	2.08	8.32E-11	9.30E-09	6.28E-02	0.04	2.51E-03
Th-228	0.74	2.96E-14	2.81E-15	8.22E-12	0.04	3.29E-13
Total U	1.4	5.6E-10	5.77E-09	1.51E-03	0.04	6.02E-05
Sr-90/Y-90	1.8	1.064E-08	1.10E-06	2.46E+00	0.04	9.8E-02
Total				2.54E+00		1.02E-01

Table 6-44. Estimated Hazard Index (HI) for Wildlife Potentially Exposed to Constituents of Concern in the 200-BP-1.

Constituent	Receptors									
	Pocket Mouse	Jack Rabbit	Swainson's Hawk		Loggerhead Shrike		Owl		Coyote	
			mice	insect	mice	insect	mice	insect	veg	mice
Hazard Index[a]	0.80	0.34	0.55	0.0014	36	0.061	15	0.033	0.0051	0.51

[a] Hazard index is organism dose rate divided by benchmark dose rate of 0.2 rad/d.

7.0 SUMMARY AND CONCLUSIONS

An RI is a complex, multiple-objective component of the environmental regulatory process. As such, it demands the use of a multi-disciplinary investigational approach to define the nature and extent of threats to human health and the environment posed by contaminant releases at a site, and to collect any other information needed to support an evaluation of remedial alternatives during the FS phase of the project. The work which has been presented in this report constitutes the initial phase (Phase I RI) of this process.

In this chapter, a summary of the findings of the 200-BP-1 operable unit Phase I RI is presented. This summary is presented below in Section 7.1. Based on these findings, and on the statutory requirements of CERCLA, the NCP and the Tri-Party Agreement, remedial action objectives for the feasibility study and recommendations for further hazardous substance response work for the 200-BP-1 operable unit are presented in Section 7.2.

7.1 PHASE I REMEDIAL INVESTIGATION SUMMARY

7.1.1 Physical Characteristics

Source Area

A detailed description of the regional and local aspects of the 200-BP-1 operable unit physical characteristics is provided in Chapter 3. The following summary focuses on the major issues related to the contaminant sources, meteorology, surface hydrology, geology, soils, hydrogeology, and ecology.

The 200 Areas were designated an NPL site by EPA in November 1989. The 200 Area NPL site is divided into waste area groups largely corresponding to the major processing plants (e.g., B Plant). Each waste area group is further subdivided into one or more operable units based on waste disposal information, location, facility type and other site characteristics. The 200-BP-1 operable unit is one of the 10 operable units within the B Plant waste area group.

The 200-BP-1 operable unit is located in the approximate center of the Hanford Site, along the northern boundary of the 200 East Area. The operable unit encompasses an area of approximately 10 ha (25 ac) with the majority of the waste management units concentrated in a 1.6 ha (4 ac) region at the eastern end of the operable unit. The 200-BP-1 operable unit waste disposal activities were associated with the management of waste from the U Plant uranium reclamation operations and waste storage condensate from the adjacent 241-BY Tank Farm (Section 3.1.1).

The 200-BP-1 operable unit contains 14 waste management units (10 inactive cribs and 4 unplanned releases):

- **Cribs.** Cribs 216-B-43 through -49 received tributyl phosphate (TBP) supernatant waste generated in the TBP process, which occurred in the 221-U building. The TBP process was used for the recovery of uranium metal

from wastes generated in the bismuth phosphate ($BiPO_4$) process in the B Plant. Waste sent to cribs 216-B-50 and -57 consisted of storage tank condensate from the in-tank solidification units nos. 1 and 2 (ITS nos. 1 and 2), respectively. Both units are located in the 241-BY Tank Farm. The tenth crib (216-B-61) was constructed, but there is no historical evidence that it was ever used or received any waste (Section 3.1.2).

The disposal cribs were designed to receive, disperse and infiltrate liquid waste effluents underground. Effluents were discharged via pipeline to engineered gravel infiltration zone usually located 3-4.6 m (10-15 ft) below ground surface. The exact concentrations and quantities of radionuclides and contaminants of concern discharged to the cribs is uncertain; however records indicate that cribs 216-B-43 through -49 received an estimated 33,840,000 l (8,940,000 gl) of tributyl phosphate supernatant waste from 1965-55, and cribs 216-B-50 and -57 received an estimated 139,200,000 l (36,777,000 gl) of in-tank solidification condensate from 1965-74 (Section 3.1.2).

- **Unplanned Releases.** Four unplanned releases have been identified within the 200-BP-1 operable unit. These unplanned releases have been designated as the following waste units:

  - UN-200-E-9
  - UN-200-E-63
  - UN-200-E-89
  - UN-200-E-110

  Little information is available regarding the details of these UNs. UN-200-E-63 consisted of surface contamination from tumbleweeds. Waste unit UN-200-E-9 involved approximately 41,635 l (11,000 gl) of TBP supernatant waste which leaked onto the ground from the 216-B flush tank. Most of the wastes were removed and covered with clean soil. Waste unit UN-200-E-89 consisted of an approximately 5.15 ha (12.7 ac) area characterized by surface radiation levels which exceeded allowable levels. Interim stabilization activities were undertaken at the UN consisting of a combination of scraping and re-placement of surface contaminated soils, followed by covering with clean soil and rock. Waste unit UN-200-E-110 involved first-cycle waste from the 112-BY tank in the 241-BY Tank Farm and impacted an area of approximately 2,320 m^2 (25,000 ft^2) around the 112-BY pit. It is possible, though currently not known, that the release may have flowed onto the 200-BP-1 operable unit (Section 3.1.2.3).

The 200-BP-1 operable unit is bordered by source-operable units 200-BP-4 to the east, 200-BP-7 and 200-BP-3 to the south, and 200-BP-10 to the west. The 600 Area borders the 200-BP-1 operable unit to the north. Groundwater beneath the 200-BP-1 operable unit is contained within a separate groundwater operable unit (Section 3.1.3).

Meteorology/Surface Water Hydrology

The 200-BP-1 operable unit is situated within an area possessing a relatively moderate semiarid climate characterized by low precipitation, high evapotranspiration, and light winds (Section 3.2). There are no perennial or ephemeral streams in the 200 East Area or adjacent land, as the topography is relatively flat and the precipitation, combined with high evapotranspiration, provides little water to generate runoff. Surface drainage from the 200-BP-1 operable unit is primarily to the north. West Lake, located approximately 2.8 km (1.7 mi) north of the 200 East Area, is the nearest surface water body to the 200-BP-1 operable unit. The pond, with a surface area of approximately 4 ha (10 ac) and a depth of approximately 1 m (3 ft), is located in a topographic depression where it intersects the surface of the water table. As such, the lake constitutes a surface representation of the water table. The source of recharge to the lake is groundwater which is locally mounded as a result of 200 Areas operations. The Columbia River, an important regional surface water resource, is located to the north and east of the 200-BP-1 operable unit. At its closest approach to the operable unit, the river is located approximately 20 km (12 mi) to the northwest (Section 3.3).

Geology

The operable unit is underlain by massive basalt flows that form a regional bedrock, and by the overlying assemblage of sedimentary deposits. The geologic units of interest in the vicinity of the 200-BP-1 operable unit include, from oldest to youngest: (1) the Pomona Member of the Saddle Mountains Basalt, (2) the Rattlesnake Ridge interbed of the Ellensburg Formation, (3) the Elephant Mountain Member of the Saddle Mountains Basalt, (4) the Ringold Formation, (5) the Hanford formation, and (6) the Holocene surficial deposits. The lower of the two basalts, the Pomona Member, varies in thickness from approximately 50 m (164 ft) throughout much of the study area to 0 m (0 ft) along the axis of the Umtanum Ridge-Gable Mountain anticline (in the Gable Gap area) where the unit has been completely eroded by floodwater erosion (Section 3.4.3.2.1). Overlying the Pomona Member is the Rattlesnake Ridge interbed, a sedimentary deposit which varies in thickness from over 25 m (82 ft) in the southwest portion of the 200 East Area to 0 m (0 ft) in the Gable Gap Area where the unit has been completely removed by erosion. The interbed consists of air fall and fluvially-reworked fine-grained siliciclastic material, and arkosic sands (Section 3.4.3.2.2).

The uppermost basalt in the immediate vicinity of the 200-BP-1 operable unit is the Elephant Mountain Basalt Member of the Saddle Mountains Basalt Formation. The Elephant Mountain Member thickens from approximately 21 m (69 ft) in the northwestern portion of the study area to over 30 m (100 ft) in the south. As with other units in the area, the buried surface of the Elephant Mountain Member has been eroded to varying degrees by floodwater erosion. North of the 200-BP-1 operable unit, in the Gable Gap Area, the unit has been completely removed, exposing underlying deposits to the unconsolidated sediments of the Hanford formation. The total depth and extent of erosion in the Gable Gap Area and the geologic materials which are exposed are uncertain. In addition, an erosional "window", which may be continuous with the erosional feature in the Gap Area, is apparently present in the vicinity of two boreholes (699-53-55 and 699-55-55) north of the 200-BP-1 operable unit. Within this erosional "window", the unconsolidated sediments of the Hanford formation are in direct contact with the Rattlesnake Ridge interbed (Section 3.4.3.2.3).

Overlying the bedrock is the Ringold Formation, a mixed deposit of fluvial sands and gravels, lacustrine muds and overbank deposits. The formation is present only in the

southern portion of the study area and is generally absent from the area immediately beneath the 200-BP-1 operable unit and to the north where the sediments of the Hanford formation directly overlie basalt or sedimentary interbeds in the basalt (Section 3.4.3.2.4).

Above the Ringold Formation is the Hanford formation, a variable mixture of boulders, cobbles, pebbles, sands and silts of glaciofluvial origin. The deposit is continuous over all of the study area except for Gable Mountain where basalt is exposed. Generally, the Hanford formation overlies basalt where the Ringold Formation is not present and overlies the Ringold Formation elsewhere. In the study area, the formation has been sub-divided into three primary stratigraphic sequences: an upper gravel, a middle sand and a lower gravel. The total thickness of the Hanford formation in the study area varies from approximately 50-100 m (160-320 ft) (Section 3.4.3.2.6). Holocene eolian deposits form a thin veneer (< 10 m [33 ft]) over the Hanford formation in the area of the operable unit. These deposits consist of very-fine-to-medium-grained sands or silty sands that were originally derived from the Hanford formation (Section 3.4.3.2.7).

Pedology

The soils of the operable unit vicinity are largely dominated by the characteristics of the parent material from which they are derived. As such, and due to limited weathering and soil formation, the soil characteristics are very similar to the properties of the Hanford formation. The moisture content of the soils is generally low (ranging from approximately 1 to 6% by weight) (Section 3.5).

Hydrogeology

There are two primary hydrostratigraphic units of interest beneath the 200-BP-1 operable unit. These include a highly conductive, upper unconfined aquifer which occurs within sediments of the Hanford and Ringold Formations, and a deeper, more moderately conductive confined aquifer which occurs in sediments of the Rattlesnake Ridge interbed (Section 3.6.2). Directly beneath the 200-BP-1 operable unit and immediately to the north (downgradient), the uppermost aquifer is generally very thin, ranging from only about 1-5.5 m (3-18 ft) in thickness. Further to the north (in the Gable Gap Area and in the erosional "window" feature), the aquifer thickens to approximately 20-30 m (65-100 ft). The vadose zone varies in thickness across the study area from approximately 75 m (245 ft) directly beneath the operable unit to between 25-50 m (82-165 ft) in the Gable Gap area. In the study area, except for localized areas, all portions of the vadose zone are comprised of sediments of the Hanford formation (Section 3.6.2.1).

On the Hanford Site, groundwater flow in the unconfined aquifer is primarily from the recharge areas, principally small ephemeral streams located around the western periphery of the Pasco Basin, to the discharge zones located along the Columbia River. Superimposed on this predominantly west to east groundwater flow pattern is an artificial pattern of flow which has arisen from the waste disposal activities associated with Hanford Site operations. In the separations area, disposal activities at B pond have resulted in the creation of a water table mound located beneath the eastern portion of the 200 East Area. Groundwater flows radially outward from this mound. In the 200-BP-1 operable unit study area, groundwater in the unconfined aquifer flows under the influence of this mound towards the Gable Gap area to the north (Section 3.6.2.2).

The uppermost regionally confined aquifer is the Rattlesnake Ridge aquifer. The Elephant Mountain Basalt Member acts as a confining layer separating the Rattlesnake Ridge from the uppermost aquifer except for in areas where the basalt has been removed by erosion. As with the uppermost aquifer, the predominant flow direction is primarily from west to east. The groundwater mound which has developed in the unconfined aquifer, however, has lead to the creation of a localized disturbance to this general pattern. The creation of the water table mound has resulted in the establishment of downward hydraulic gradients between the uppermost and confined aquifers, and has lead to a radial pattern of flow beneath B-Pond in the Rattlesnake Ridge aquifer. North of the 200 East Area, flow in the Rattlesnake Ridge aquifer is apparently towards the Gable Gap Area. Potentiometric data suggests that once reaching the Gap Area, discharge from the Rattlesnake Ridge aquifer is to the uppermost aquifer. Groundwater of the unconfined aquifer north of Gable Gap then flows generally to the east where it discharges to the Columbia River (Section 3.6.2.3).

Currently, the downward gradients between the two aquifers are confined to the immediate vicinity of B pond. In the past, however, the area characterized by downward gradients is expected to have been significantly larger in areal extent. In conjunction with erosional features and possibly joint/fracture systems, the introduction of contaminants to the Rattlesnake Ridge confined aquifer occurred. There are at least one, and possibly two, erosional features where direct aquifer communication has occurred in the past (Section 3.6.2.4). Currently however, except for the B pond area, gradients between the two aquifers are generally upward throughout the study area (Section 3.6.2.4).

Ecology

For reasons of national security, as well as to insure public health and safety, access to the entire Hanford Site has been administratively controlled since 1943. Access to most of the 200 Areas, including the 200-BP-1 operable unit, is restricted in addition to the general Hanford Site administrative controls. Although DOE is expected to retain control of the Hanford Site for the foreseeable future, a Hanford Future Site Uses Working Group (1992) has proposed land use alternatives for DOE to consider when it develops an environmental impact statement for the Hanford Site (Section 3.7.1.1.2).

Since 1944, there has been no agricultural activity or resident human population on the Hanford Site. Based on 1980 census data, approximately 110 people live within 17 km (10 mi) of the 200 East Area. The nearest resident is at least 12 km (7 mi) away from the 200-BP-1 operable unit as this is the approximate distance to the nearest Hanford Site boundary. The working population for all 200 Areas shifts is approximately 2,400 (Section 3.7.1.1.2).

The Columbia River, which is located approximately 12 km (7 mi) from the 200-BP-1 operable unit at its closest approach, is the most significant surface-water body in the region and is used as a source of drinking water, industrial process water, crop irrigation, and for a variety of recreational activities, including fishing, hunting, boating, water skiing, and swimming (Section 3.7.1.2.1). Downgradient groundwater in the immediate vicinity of the 200-BP-1 operable unit is not used for either drinking or irrigation. The nearest drinking water supply wells are those that serve the 400 Area, approximately 13 km (8 mi) to the southeast of the operable unit (Section 3.7.1.2.2), and not hydraulically downgradient of the 200-BP-1 operable unit.

The 200-BP-1 operable unit is located on the 200 Areas plateau. Habitat within the undisturbed areas of the 200 Areas plateau consist primarily of the big sagebrush/Sandberg's bluegrass community that dominates most of the Hanford Site. Because of the operational disturbances which have taken place in the operable unit, however, the majority of the plants occurring in the 200-BP-1 operable unit are invader species. In the nonradiation areas the major vegetation includes rabbitbrush, Sandberg's bluegrass, Russian thistle and cheatgrass. In the radiation zone plant species include revegetated wheatgrass and thickspike used to stabilize soil of two cribs. The major plants around the cribs are cheatgrass, Sandberg's bluegrass, Russian thistle and rabbitbrush. No endangered or threatened plant species were observed in the operable unit (Section 3.7.2.1).

The most abundant fauna in the region include the western meadowlark, Great Basin pocket mouse, deer mouse, Townsend's ground squirrel, Black-tailed jackrabbit, mule deer, coyote, badger, and various raptor and insect species. The ferruginous hawk, Swainson's hawk, loggerhead shrike, and long-billed curlew are candidate species for inclusion on the Federal threatened or endangered species list. Candidates for state listing are the burrowing owl, sage sparrow, and sage thrasher. With regards to mammals, two species of concern are Merriam's shrew, a candidate for listing by Washington State, and the Pacific western big-eared bat, a candidate for federal protection. None of these candidate species were observed on the 200-BP-1 operable unit, however, they are commonly associated with the big sagebrush/Sandberg's bluegrass habitat of the 200 Area plateau. There are no Federal- or State-classified threatened, endangered or candidate species among the reptiles and insects of the Hanford Site.

7.1.2 Nature and Extent of Contamination

The 200-BP-1 operable unit includes ten inactive cribs (known as the 216-B cribs) and four unplanned releases. Crib (216-B-61) was constructed, but there is no evidence that it was ever used or received any wastes. In Chapter 4, a step-wise screening process is presented which defines the extent of contamination associated with these waste management units. In this screening process, chemical constituents detected in soil and groundwater samples are compared to levels observed in sample blanks, established background concentrations, and to calculated risk-based screening levels with the goal of identifying those compounds that constitute actual contamination and pose risk to human health and the environment. These compounds which are defined in this process are designated contaminants of potential concern, and are retained for later use in the baseline risk assessment performed in Chapter 6. Contaminants of potential concern for groundwater will not be carried forward to Chapter 6 as the risk assessment will not be performed for the groundwater pathway in this RI. All compounds which are eliminated by the screening process are dropped from further consideration in this RI. This subsection will summarize the chemical and contaminant characteristics of the operable unit which result from this screening process (Sections 4.1 and 4.2).

Risk-based screening was conducted for soil and groundwater media. For soils, the screening was performed separately for two different zones. The first zone, near surface soils, is defined as soil between the ground surface and 4.6 m (15 ft). This zone consists of stabilized surface soils, trench backfill and the bottom 0.3-1.2 m (1-4 ft) of crib infiltration gravels. The second zone consists of soils beneath 4.6 m (15 ft). This zone, designated subsurface infiltration gravels/soils, includes the remainder of the crib infiltration gravels

(approximately 1.5 m [5 ft]) and underlying natural materials. The 4.6 m (15 ft) cut-off is based on guidelines presented in MTCA. For both the near-surface soils and subsurface infiltration gravels/soils, three soil exposure pathways are used in calculating preliminary risk-based benchmark concentrations: soil ingestion, air inhalation (including inhalation of fugitive dust) and external exposure to radioactivity. For groundwater, the only exposure mechanism evaluated is groundwater ingestion (Section 4.3).

7.1.2.1 Soils. For near surface soils, the identified contaminants of potential concern include the following (Section 4.3.2):

- cesium-137, radium-226, strontium-90, thorium-228, and total uranium.

For subsurface infiltration gravels/soils, the identified contaminants of potential concern include the following:

- cadmium, nickel, tributyl phosphate, PCB, antimony-125, cesium-137, cobalt-60, plutonium-238, plutonium-239, plutonium-239+240, radium-226, strontium-90, technetium-99, thorium-228, and total uranium.

Although contamination in soils was detected at depths of up to 71.9 m (236 ft), maximum radionuclide concentrations were generally observed in the 4.6-15.2 m (15-50 ft) depth range. This depth interval represents the base of the crib gravel and underlying native soil where the observed contaminant distributions are consistent with the relative immobility of many of the radioactive constituents. Below 15.2 m (50 ft), levels generally decline until a depth of approximately 30 m (100 ft) at which concentrations remain uniformly low. Soils above 3.7-4.6 m (12-15 ft) are characterized by relatively low radionuclide levels, as compared to the crib gravels and deeper zones. Results of the soil sampling and spectral gamma-ray logging indicate that contamination is generally confined to the area beneath the cribs and that significant lateral waste migration due to perched groundwater conditions does not appear to have occurred (Section 4.4.1).

7.1.2.2 Groundwater. For groundwater, the identified contaminants of potential concern include the following (Section 4.3.3):

- antimony, arsenic, barium, cadmium, total and free cyanide, complexed cyanide, chromium, copper, lead, manganese, nickel, selenium, silver, thallium, vanadium, fluoride, nitrate, nitrite, sulfate, trichloroethene, 4,4'-DDT, cobalt-60, potassium-40, plutonium-238, radium-226, radium-228, strontium-90, technitium-99, total uranium and tritium.

Extensive contamination of groundwater in the 200-BP-1 operable unit study area has occurred due to Hanford Site operations. This contamination is likely the result of multiple sources located throughout the 200 East and West Areas. While still in exceedance of regulatory standards, levels which are currently present in groundwater are, for most radioisotope constituents, one or more orders of magnitude less than concentrations which occurred in the early years of Hanford operations (Section 4.4.2.1).

Contaminant plumes which are present throughout the study area at concentrations significantly in exceedance of background levels, risk-based screening concentrations, or other regulatory criteria were identified for gross beta, total cyanide, cobalt-60, nitrate, technetium-

99, and tritium. For the other contaminants of potential concern in groundwater, the extent of contamination tends to be confined to localized areas and/or is characterized by relatively lower concentrations. Generally, the plumes are centered at well 699-50-53a (located about one (1) km north of 200-BP-1) where maximum contaminant levels are consistently observed. Well 699-55-57 is also characterized by relatively high concentrations as compared to other monitoring wells throughout the study area (Section 4.4.2.2).

7.1.3 Contaminant Fate and Transport

Chapter 5 consists of a transport analysis of the contaminants of potential concern identified in Chapter 4. The purpose of this analysis is to evaluate the behavior of each contaminant of potential concern in the environmental media in which it is transported, specifically to provide estimates of the contaminant concentrations at points of potential receptor exposure. Pathways which were evaluated include air, surface water, subsurface water, and biota. Although the subsurface water pathway includes both an unsaturated and a saturated pathway, transport in the saturated pathway is not addressed in this RI since the groundwater portion of this operable unit has recently been removed as per Change Order M-15-92-5 dated December 3, 1992. The saturated groundwater pathway will be addressed in the 200 East Aggregate Groundwater Study.

Potentially operative contaminant transport pathways which were evaluated for the 200-BP-1 operable unit, taking preliminary toxicity screening results of Chapter 4 into account, include the following:

- Air emissions of Rn-222 and Rn-220 from the near surface and subsurface infiltration gravels/soils: These radioisotopes are gaseous decay products of Ra-226, and Th-228, respectively. Because of the relatively long half-life of Ra-226, and the small quantities of both Th-228 and Ra-226 present in the crib soils, radioactive decay is not considered a significant airborne release mechanism for this operable unit (Section 5.2.1.1).

- Volatile emissions of PCBs from the subsurface infiltration gravels/soils: Because of the low vapor pressures and high soil adsorption constants characteristic of PCBs, as well as the low PCB concentrations measured, the volatilization release mechanism is not considered to be significant at this operable unit (Section 5.2.1.2).

- Fugitive dust emissions and atmospheric dispersion of non-volatile compounds cesium-137, radium-226, strontium-90, thorium-228, and total uranium from the near surface soils; cadmium, nickel, tributyl phosphate, PCB, antimony-125, cesium-137, cobalt-60, plutonium-238, plutonium-239, plutonium-239+240, radium-226, strontium-90, technetium-99, thorium-228, and total uranium from the subsurface infiltration gravels/soils:

  Fugitive dust emission and subsequent atmospheric transport was modeled using EPA's Fugitive Dust Model (FDM). The FDM is a computer code which analytically solves a commonly used air dispersion equation. The model was used to calculate concentrations of fugitive dust and the resulting fallout which will characterize the study area in the year 2018

(2018 represents the earliest year for which alternative land uses may occur at the Hanford Site). It was assumed in the model simulations that the current protective clean soil cover (from the interim stabilization activities) is not maintained in the future and that the contaminated near surface soils are exposed and accessible to wind erosion. Furthermore, a major disturbance of the crib infiltration gravel materials is assumed to potentially occur, such as excavation, thereby exposing these soils directly to wind erosion. These assumptions are felt to provide a strong conservative bias to the modeling results.

The results of the modeling indicate that airborne particulate matter is transported and deposited over an elliptically shaped area which extends principally to the southeast from the operable unit, and reaches the approximate vicinity of B-Pond at its furthest downwind extent. Concentrations of airborne dust vary from about 1 to $1x10^{-6}$ $\mu g/m^3$. Based on these estimates of airborne particulate concentrations and appropriate decay factors, the projected soil concentrations resulting from contaminants originating from the 200-BP-1 operable unit are calculated (Section 5.2.1.3).

- Surface water transport due to run-off from storm events and deposition of fugitive dust on nearby surface water bodies with subsequent transport: After considerations of the potential for run-off and review of the results of the fugitive dust modeling, the potential for transport by the surface water pathway was considered to be insignificant (Section 5.2.2).

- Vadose zone transport (Unsaturated Pathway Modeling) of TBP, Cs-137, Co-60, Sr-90, Pu-238, Pu-239, Pu-239/240, total U, and Tc-99 from the crib soils to the groundwater: Contaminant migration through the vadose zone, due to historical crib operation and subsequent infiltration of meteoric water, was modeled using a finite difference computer code (PORFLO-3) to identify potential future impacts to the unconfined aquifer. Other contaminants of potential concern, cadmium, nickel, PCB, Ra-226, Sb-125, Th-228 and stable daughter products of the above modeled contaminants were not modeled, because of immobility, low concentrations and/or short half-lives. These contaminants are not expected to reach the water table in amounts that would significantly impact the aquifer.

Each simulation was run until either the contaminant reached the water table or the simulations indicated that it was unlikely that the contaminant would reach the water table at significant concentrations. In general, those contaminants which were both immobile and short-lived never reached the water table at unacceptable levels. These included Cs-137 and Sr-90 which were bound in the upper 23 m (74 ft) at the time discharge to the cribs ceased. Radioactive decay following cessation of discharge prevented either contaminant from reaching the water table.

Migration of Co-60 to the water table was shown to occur rapidly with concentrations in the saturated zone reaching maximum values soon after crib operations began. Because of its short half-life, however, within 100 years after cessation of effluent discharges (≈2050) the operable unit will not

impact the aquifer with Co-60 above 1 pCi/L. The modeling results indicate that the most mobile constituents, Tc-99 and nitrate, have reached the aquifer at maximum concentration shortly after cessation of discharge to the cribs. This is consistent with observed groundwater monitoring results (Appendix E). Residual Tc-99 and nitrate are currently entering and impacting the groundwater at concentrations of about 400 - 700 pCi/L and 10 - 20 mg/L, respectively.

Migration of TBP to the water table was estimated to occur but would probably take over 1000 years. TBP is an organic compound and is expected to degrade but its actual degradation rate is unknown. TBP will likely degrade to some degree before reaching the saturated zone. Since TBP is present at relatively low concentrations (95% UCL of 16,000 μg/Kg) in the infiltration gravels/soils and will likely degrade before reaching the water table, future impacts from this operable unit are expected to be insignificant.

Of the compounds modeled, future potential impacts to groundwater were confined to plutonium and uranium. Although plutonium has a distribution coefficient in the same order-or-magnitude to that of Cs-137, it also has an extremely long half-life thereby allowing it to reach the water table in an estimated 9,500 yr. Diffusion is expected to be the dominant transport mechanism for plutonium. Insufficient computer resources prevented prediction of the maximum concentration of plutonium in groundwater, although it is expected to be significant (>100 pCi/L), but will take tens-of-thousands of years. Total U is shown reaching the water table by about 15 yr after the start of crib operations. The maximum total U concentrations in the saturated zone could exceed 3,000 pCi/L approximately 4,500 years in the future.

- Terrestrial biological transport of cesium-137, radium-226, strontium-90, thorium-228, and total uranium from near surface soils to Swainson's hawk, loggerhead shrike, and burrowing owl: Two generalized contaminant transport pathways for the species of concern were considered. The first model assumed that the exposure mechanism for the birds was ingestion of the Great Basin pocket mouse, while in the second model the bird's diet consisted entirely of insects. Within each type of prey, estimated contaminant concentrations attributable to the operable unit were estimated for the near surface soil contaminants of potential concern. Due to the lack of species-specific concentration data, conservative simplifying assumptions and surrogate uptake factors were necessary in the biological transport pathway evaluation.

7.1.4 Risks to Human Health and the Environment

Human Risk Assessment

The human health baseline risk assessment evaluates the risks posed by contaminants in the 200-BP-1 operable unit under four exposure scenarios (industrial, residential,

recreational, and agricultural) and three locations (operable unit, Hanford Site, off Hanford Site). These evaluations are performed for current conditions as well as for several future conditions. Evaluation of the future conditions is divided into three possibilities (see Section 6.2.1.1):

- The clean soil cover is maintained and undisturbed
- The near surface soils are exposed and/or excavated
- The infiltration gravels/soils are exposed and deposited on the ground surface.

Non-radioactive contaminants were evaluated for both non-carcinogenic and carcinogenic effects, as appropriate. Radioactive contaminants were evaluated only for their carcinogenic potential. No systemic toxic effects are expected to occur as a result of exposure to contaminants at the operable unit under any scenario; the largest estimated HQ is 0.007. The remaining summary of the human health assessment focuses on estimates of cancer risk (Section 6.2.3.1).

Evaluation of the current scenario recognizes that a clean soil cover currently exists at the operable unit and that access to the unit is currently limited. For both crib groupings (cribs 216-B-43 through -50 and crib 216-B-57), the estimated cancer risks under the current scenario is less than 10^{-6} (Section 6.2.3.2.1).

Environmental monitoring data are used to estimate cancer risks associated with contaminants attributable to sources other than the 200-BP-1 operable unit. Air monitoring data from the 200 Area represent ambient air concentrations of constituents to which workers on the operable unit are exposed. External radiation data from distant communities represent naturally-occurring sources and general anthropogenic sources (e.g., worldwide fallout from atmospheric nuclear bomb testing) of external exposure to which operable unit workers are exposed. The risks associated with these exposures indicate that the total risk ($2x10^{-4}$) incurred by workers on the operable unit is dominated by external exposure to naturally-occurring radionuclides in the soil, and that the current risk associated with operable unit contaminants is three orders of magnitude smaller than this total (Section 6.2.3.2.1).

Although future land use of the 200 Area is assumed to be limited to industrial activities, for the purposes of this human health baseline risk assessment radiological controls are assumed to be nonexistent for the future industrial scenarios. Even without restricting access to the operable unit, this risk assessment indicates that a cancer risk of 10^{-6} is exceeded only under the future industrial scenario for receptors on the operable unit, and only if near surface soils or infiltration gravels/soils are uncovered, permitting direct contact with contaminants (Section 6.2.3.2).

Assuming near surface soils are exposed, the largest risk for operable unit receptors is about $1x10^{-4}$. Risk estimates associated with excavation of infiltration gravels/soils (which are approximately 4.6 m (15 ft) below ground surface) for operable unit receptors exceed 10^{-2} for both crib groupings. Because risk estimates made using a linear dose-response equation become increasingly inaccurate as they approach a value of 1, LICR values that exceed 10^{-2} are reported as $>10^{-2}$ (Section 6.2.3.2.2).

Risk estimates for receptors on the Hanford Site (excluding the 200 Area) are all less than 10^{-6} under all future conditions. Estimates associated with infiltration gravels/soils

include the contribution of exposures resulting from fugitive dust deposition of cesium-137 and strontium-90 (Section 6.2.3.2).

The risk estimates for future conditions do not consider the probability that the clean soil cover will be absent or present in the year 2018, or that future operable unit workers will excavate the near surface soils or infiltration gravels/soils. In addition, because this is a deterministic risk assessment, the uncertainty associated with all risk estimates cannot be quantified. In order to compensate for the uncertainty associated with input parameters, estimates used to characterize these parameters are often conservatively biased. As a result, the risk estimates provided in this assessment represent a set of assumptions which, as a whole, is extremely unlikely. Use of a more realistic set of assumptions is likely to yield significantly lower risk estimates (Section 6.2.3.3).

Ecological Risk Assessment

The baseline ecological risk assessment evaluates the impact of near surface soil contaminants on six indicator species (Great Basin Pocket mouse, Jack rabbit, Swainson's hawk, loggerhead shrike, burrowing owl, and coyote). No data have been obtained to provide concentrations of contaminants in biological media within the operable unit; therefore, biological uptake of contaminants by indicator species is estimated by combining soil concentrations, bioconcentration factors, gastro-intestinal absorption efficiencies, biological half-lives, and food intake rates. Inhalation and soil ingestion pathways were not evaluated. External exposure to wildlife from radioactive contaminants has been shown to be a minor contributor to dose and was not evaluated (Section 6.3).

The ecological assessment is based on the assumption that a clean soil cover does not exist, and that no controls exist to prevent access to the operable unit by the indicator species. In addition, it is assumed that ecological conditions on the unit (e.g., vegetation and food supply) are identical to the natural habitat of these species. Such conditions do not currently exist, and, given the assumption of permanent industrial land use in the 200 Area, the presence of indicator species on the operable unit could be controlled indefinitely.

The frequency of exposure of a receptor to contaminants was determined by estimating the proportion of the site area to the receptor's home range. For organisms whose home range is smaller than the operable unit, it is assumed that 100% of their diet consists of contaminated foodstuffs. The 95% UCLs of the mean soil concentrations are used as the exposure concentrations for modeling and dose calculations (Section 6.3.2).

Contaminant intakes are calculated in units of mg/kg-d for non-radioactive contaminants and rad/d for radionuclides. These intakes are divided by benchmark intakes to yield (unitless) hazard indices. The benchmark intake level for radioactive contaminants is 0.2 rad/d (20% of the 1 rad/d dose rate at which slight effects of radiation become apparent in important population maintenance attributes). A HI value greater than 1 indicates that a benchmark intake value may be exceeded, potentially resulting in an adverse impact on the ecological receptor population (Section 6.3.3).

For the six key receptors evaluated, HI values range from 0.001 to 16 for radioactive contaminants. The potentially impacted receptors (HI > 1) are the Pocket mouse, Jack rabbit, loggerhead shrike, and burrowing owl. This assessment indicates that these receptors could be adversely impacted by near surface soil contaminants on the operable unit if exposure to

these soils is allowed to occur. Uncertainty associated with the results of the ecological assessment are extremely high because of the limited number of exposure pathways evaluated, and the conservative assumptions made in the evaluation of the food chain pathway (Section 6.3.3).

7.2 PHASE I REMEDIAL INVESTIGATION CONCLUSIONS AND RECOMMENDATIONS

In this section, the results of the Phase I RI for the 200-BP-1 operable unit, as summarized above in Section 7.1, are used to develop recommendations for conducting and focusing further hazardous substance response activities needed to comply with the terms of CERCLA, the NCP, and the TPA. Section 7.2.1 consists of a review of potential federal and state applicable or relevant and appropriate requirements (ARARs) which may be pertinent to the remedial actions which are considered in the FS. Identification and refinement of ARARs is an ongoing process which occurs throughout all phases of the RI/FS. The review of ARARs included herein is an update of the preliminary ARAR identification which was performed during scoping of the work plan. Remedial alternative evaluation in the FS should be performed in the context of compliance with the federal and state regulations that are ARARs. Section 7.2.2 provides recommendations for expedited response actions in accordance with the process outlined in 40 CFR 300. Section 7.2.3 recommends remedial action objectives for use in the early stages of the FS for the 200-BP-1 operable unit. Section 7.2.4 provides recommendations for Phase II RI activities, in terms of additional operable unit characterization.

7.2.1 Applicable or Relevant and Appropriate Requirements

This sub-section presents potential applicable or relevant and appropriate requirements (ARARs) for the 200-BP-1 operable unit based on information collected during the Phase 1 remedial investigation. This section identifies and evaluates federal and state requirements which are applicable or relevant and appropriate chemical and location-specific requirements to the 200-BP-1 operable unit. Action-specific ARARs will be presented in the 200-BP-1 Feasibility Study. The ARAR development process is based on CERCLA guidance (EPA 1988a and EPA 1988b).

Section 121 of the Comprehensive Environmental Response, Compensation and Liability Act, as amended establishes cleanup standards for remedial actions under CERCLA. Section 121 requires in part that, any applicable or relevant and appropriate standard, requirement, criteria or limitation under any federal environmental law, or any more stringent state requirement promulgated pursuant to a state environmental statute, be met for any hazardous substance, pollutant, or contaminant remaining on-site.

A requirement for Superfund compliance at a hazardous substance cleanup site may be either "applicable" or "relevant and appropriate", but not both. Identification of ARARs must be done on a site-specific basis and involves a two-part analysis: first, a determination is made whether a given requirement is applicable; then if it is not applicable, a determination is made whether it is nevertheless both relevant and appropriate. EPA guidance also includes To-Be-Considered (TBC) materials which are advisories and non-promulgated guidance issued

by federal or state governments that are non-statutory requirements evaluated along with ARARs as part of the risk assessment used to establish protective cleanup limits.

The EPA may waive ARARs and select a remedial action that does not attain the same level of cleanup as identified by ARARs. Section 121 of the Superfund Amendments and Reauthorization Act identifies six circumstances where EPA may waive ARARs for on-site remedial actions.

The six circumstances are:

- The remedial action selected is only a part of a total remedial action (such as an interim action) and the final remedy will attain the ARAR upon its completion.
- Compliance with the ARAR will result in a greater risk to human health and the environment than alternative options.
- Compliance with the ARAR is technically impracticable from an engineering perspective.
- An alternative remedial action will attain an equivalent standard of performance through the use of another method or approach.
- The ARAR is a state requirement that the state has not consistently applied (or demonstrated the intent to apply consistently) in similar circumstances.
- Section 104, Superfund-financed remedial actions, compliance with the ARAR will not provide a balance between protecting human health and the environment and the availability of Superfund money for response at other facilities.

There are several different types of requirements that CERCLA actions may have to comply with and are generally identified as chemical-specific, location-specific and action-specific ARARS. The following definitions are excerpts from EPA guidance in *CERCLA Compliance with Other Laws Manual: Interim Final* (EPA 1988b). However, some requirements may not fall neatly into the classification system.

Chemical-specific requirements are usually health- or risk-based numerical values or methodologies which, when applied to site-specific conditions, result in the establishment of numerical values. These numbers establish the acceptable amount or concentration of a chemical that can be found in, or discharged to the ambient environment.

Location-specific requirements are restrictions placed on the concentration of hazardous substances or the conduct of activities because they occur in special or sensitive locations or environments.

Action-specific requirements are those that place either technology-based or activity-based requirements on remedial actions at CERCLA sites.

Federal and state regulations along with other guidance were evaluated as potential ARARs and TBC materials. The laws and regulations evaluated a potentially applicable or relevant and appropriate requirements for the 200-BP-1 operable unit are summarized in Tables 7-1 and 7-2. Significant requirements are discussed in the following sections.

7.2.1.1 Chemical-Specific ARARs. Chemical-specific ARARs may be federal, state regulations and other guidance that identify acceptable health or risk based contaminant levels for different media known to be contaminated.

7.2.1.1.1 Federal Chemical-Specific ARARs. Federal chemical-specific requirements, criteria, or guidance for the contaminants of concern at the 200-BP-1 operable unit are listed in Table 7-1. Regulatory limits for contaminants of concern, established by these regulations are presented in Tables 7-3 and 7-4.

National Primary Drinking Water Regulations

Requirements of the National Primary Drinking Water Regulations (40 CFR 141) promulgated under the Safe Drinking Water Act (SDWA) are not applicable to the 200-BP-1 operable unit. However, they would be relevant and appropriate requirements to the 200-BP-1 operable unit if the site converts to other land uses. The regulations are applicable to contamination in community water systems. The regulations establish maximum contaminant level goals (MCLGs) and maximum contaminant levels (MCLs). MCLs and MCLGs are for both non-radioactive contaminants and radionuclides. Remedial alternatives must consider protection of potential drinking water supplies as required by Section 300.430 (e)(2) of the National Oil and Hazardous Substances Contingency Plan (NCP). The NCP requires that remedial actions for ground or surface water that are current or potential sources of drinking water shall attain standards established under the SDWA. Groundwater affected by the 200-BP-1 operable unit is not currently used for drinking, however it could be used in the future if the site is released from institutional controls. There is also potential for discharge of contaminated groundwater to the Columbia River which is used for drinking water. Remedial alternatives for the 200-BP-1 cribs and contaminated soils need to evaluate actions to prevent migration of contaminants from soils to groundwater at concentrations that cause the groundwater to exceed MCLGs and MCLs. Drinking water MCLGs and MCLs for contaminants of concern are listed in Tables 7-3 and 7-4.

National Secondary Drinking Water Regulations

The National Secondary Drinking Water Regulations control contaminants in drinking water that primarily affect aesthetic qualities of the water that relate to public acceptance. These regulations are not applicable nor relevant and appropriate to the 200-BP-1 operable unit since they are not federally enforceable. However, under Washington State regulations (173-340-720(2)(9)(ii)) they are a potential ARAR because State regulations specify secondary maximum contaminant levels (SMCLs) as enforceable standards. Secondary maximum contaminant levels established for operable unit contaminants of concern are presented in Table 7-4.

National Emission Standards for Hazardous Air Pollutants

EPA standards for radionuclides from facilities owned and operated by DOE under 40 CFR 61.90, National Emission Standards for Hazardous Air Pollutants are potentially

applicable since radionuclides are present in operable unit soils and there is potential for airborne release.

Environmental Radiation Protection Standards for the Management and Disposal of Spent Nuclear Fuel, High-Level and Transuranic Radioactive Waste

Environmental Radiation Protection Standards for the Management and Disposal of Spent Nuclear Fuel, High-Level, and Transuranic Radioactive Waste (40 CFR 191) are potentially applicable requirements to the 200-BP-1 operable unit since the standard applies to Department of Energy facilities. Groundwater protection standards established under the regulation specify that disposal systems provide a reasonable expectation that for 1,000 years after disposal, the system shall not cause the radionuclide concentrations averaged over any portion of a special source of groundwater to exceed the levels presented in Table 7-3. However, the proposed rule published on February 10, 1993 (58 FR 7924) proposes that disposal systems shall be designed so that for 10,000 yr of undisturbed performance after disposal shall not cause the levels of radioactivity to exceed the limits specified in 40 CFR 141, as they exist on the date the implementing agency determines compliance. Table 7-3 summarizes the maximum allowable exposure to members of the public that may result from activities conducted at these facilities.

Health and Environmental Protection Standards for Uranium and Thorium Mill Tailings

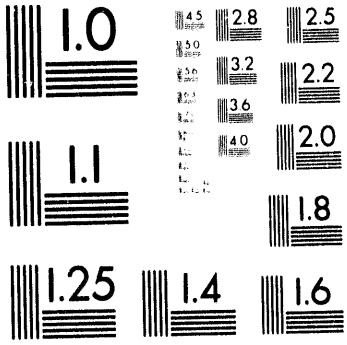
Requirements of 40 CFR 192, Health and Environmental Protection Standards for Uranium and Thorium Mill Tailings are potentially relevant and appropriate requirements to the operable unit since they establish maximum acceptable cleanup levels for Ra-226, 228 and gross alpha radioactivity, which are contaminants of the 200-BP-1 operable unit. The standard states that remedial actions intended to stabilize the site be designed to be effective for up to 1,000 years, to an extent reasonably achievable, and in any case, no less than 200 years. This is a design standard and monitoring after disposal is not required to demonstrate compliance. The requirements are not applicable since the facility is not associated with uranium or thorium milling.

Standards for Protection Against Radiation

NRC Standards for Protection Against Radiation found in 10 CFR 20 are relevant and appropriate to the operable unit since the regulation establishes standards for protection against radiation hazards that may result from occupational exposure or discharges to air and water. The standard is not applicable because it only applies to operations licensed by the Nuclear Regulatory Commission. The concentration limits for radionuclides in airborne and liquid effluent discharged to unrestricted areas established under the standard are summarized in Table 7-3.

DOE Order 5400.5 Radiation protection of the Public and Environment

Radiation protection and radioactive waste management requirements issued under the Atomic Energy Act are implemented at DOE facilities as DOE orders. Under CERCLA these standards are To Be Considered for activities conducted at the 200-BP-1 operable unit since they are not formally promulgated regulations. However, compliance with DOE Orders is required at Hanford. DOE Order 5400.5, Radiation Protection of the Public and Environment, includes derived concentration guides (DCGs) for radioactively contaminated liquid


1.0
1.1
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2.5
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3.2
3.6
4.0
4.5
5.0

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discharges to surface waters, aquifers, soil and sanitary sewage systems. The order incorporates most of the same control and cleanup provisions of EPA's 40 CFR 192 and provides factors used to estimate external and internal doses received from exposure to radiation, as well as expanded requirements and guidance on environmental monitoring.

7.2.1.1.2 State of Washington Chemical-Specific ARARs. CERCLA 121(d) requires that, in addition to satisfying Federal ARAR's, any state standard, requirement, criterion, or limitation that is more stringent must also be met. State requirements must be legally enforceable regulations or statutes, identified in a timely manner, and be of general applicability to all circumstances covered by the requirement. Table 7-2 identifies preliminary chemical-specific Washington State ARARs for the 200-BP-1 operable unit. Tables 7-3 and 7-4 present regulatory limits for operable unit contaminants of concern.

Model Toxics Control Act Cleanup Regulation

Regulations under chapter 173-340 WAC, which implement requirements of the Model Toxics Control Act (MTCA) are applicable to the 200-BP-1 operable unit. These regulations establish administrative processes and standards to identify, investigate and cleanup facilities where hazardous substances have been released. The state regulations have the potential to be stricter than federal standards. For example, MTCA specifies secondary MCLs as applicable requirements. Secondary MCLs are nonenforceable standards under 40 CFR 143 and are based on non-human health-based goals relating to qualities of taste and odor.

The MTCA regulations establish three basic methods for determining cleanup levels for soils, as set forth in WAC 173-340-700. These include Method A - Tables, Method B - standard method, and Method C - Conditional method. The MTCA regulations specify procedures for establishing levels that are supposed to be protective of human health and the environment based on reasonable maximum exposure assuming either a residential site use (WAC 173-340-740) or industrial site use (WAC 173-340-745). Section 740 establishes standards under all three methods and Section 745 uses only Methods A and C.

Method A is generally used for routine cleanups with relatively few contaminants. Standards for Method A cleanups are set at other federal or state ARARs, are established in Table 1 (based on residential site use) in WAC 173-340-740 and Table 3 (based on industrial site use) in WAC 173-340-745, or are based on natural background concentrations or practical quantification limits (PQLs). Since the cleanup at the 200-BP-1 operable unit cannot be considered routine and most of the contaminants are not included in Tables 1 or 3, Method A cleanup standards are not appropriate for use.

Method B is the standard method for determining cleanup levels and assumes a residential site use. Method B levels for soil are determined using federal or state ARARs or are based on risk equations specified in WAC 173-340-740. For individual carcinogens, the cleanup levels are based on the upper bound of the excess lifetime cancer risk of one in one million (1×10^{-6}). Total excess cancer risk under Method B for multiple substances and pathways cannot exceed one in one hundred thousand (1×10^{-5}). Residential use of the 200-BP-1 operable unit is not a likely scenario either currently or in the future. However, Method B soil clean up levels are shown in Tables 7-3 and 7-4 for comparison purposes.

Method C industrial cleanup levels are appropriate for use at the 200-BP-1 operable unit. Method C cleanup levels are used where; Method A or B cleanup levels are below area

background concentrations, cleanup to Method A or B levels has the potential for creating greater overall threat to human health and the environment than Method C, cleanup to Method A or B is not technically possible, or the site meets the definition of an industrial site. Method C cleanups must comply with other federal or state ARARs, must use all practical levels of treatment and must incorporate institutional controls as specified in WAC 173-340-440. Risk based equations for Method C cleanup levels for soil are specified in WAC 173-340-740 for residential and WAC 173-340-745 for industrial exposure assumptions. Total excess cancer risk for Method C cannot exceed 1 in one hundred thousand (1×10^{-5}). Method C industrial cleanup levels are most appropriate for use at the 200-BP-1 operable unit based on current and projected future land use. An industrial land use with institutional controls is currently planned for the 200 Areas, which also includes a buffer zone (Hanford Future Site Uses Working Group 1992).

All three MTCA methods for determining cleanup levels require minimum compliance with other federal or state ARARs, and consideration of cross-media contamination. For example, soil cleanup standards may be based on protection of groundwater. The regulations specify soil cleanup to levels equal to or less than one hundred times the groundwater cleanup levels established in accordance with WAC 173-340-720 unless it can be demonstrated that a higher soil concentration is protective of groundwater at the site. Vadose zone fate and transport modeling has been used at the 200-BP-1 operable unit to determine the potential of hazardous substances in the soil to impact groundwater. Groundwater cleanup levels based on WAC 173-340-720 Methods B and C are included in Tables 7-3 and 7-4 for comparison with the results of the vadose zone fate and transport modeling.

The point of compliance for soil cleanup based on protection of groundwater and for cleanup levels based on human exposure via direct contact are defined under MTCA. The point of compliance is defined as the point or points throughout the site where cleanup levels are established in accordance with the cleanup requirements for groundwater and soil specified in Sections 173-340-720 through 750. The point of compliance for soil cleanup levels based on the protection of groundwater are to be achieved in all soils throughout the site. For soil cleanup levels based on human exposure via direct contact, soil cleanup levels are established for the upper fifteen feet, from the ground surface to fifteen feet. These depths represent the extent that could be excavated or disturbed as a result of site development. For alternatives that involve containment of hazardous substances, cleanup levels typically will not meet points of compliance established for groundwater protection or human exposure via direct contact. In these cases, compliance monitoring and other requirements identified in 173-340-360 (8) may be required to ensure long term integrity of the containment system.

Department of Health Radiation Protection Standards

Radiation Protection Standards were developed pursuant to the Atomic Energy Act of 1954, as amended. The standards set maximum allowable annual radiation doses in restricted and unrestricted areas, set maximum dose levels that minors may receive in restricted areas and establish acceptable concentration limits in effluent released to unrestricted areas. Radiation protection standards contained in Chapters 220 through 255 of WAC 246 are not specifically applicable to the operable unit, however, they are relevant and appropriate because they address similar contaminants and establish standards for acceptable levels of exposure.

7.2.1.2 Location-Specific ARARs. Location-specific ARARs at the 200-BP-1 operable unit are restrictions placed on the concentration of hazardous substances or the conduct of activities at the site based solely on site characteristics of the 200-BP-1 operable unit.

7.2.1.2.1 Federal Location-Specific ARARs. Federal location-specific requirements that were evaluated are summarized in Table 7-1.

The National Historic Preservation Act

The National Historic Preservation Act of 1966 is not an ARAR since no facilities located in the operable unit are listed or considered for inclusion on the National Register of Historic Places.

The Archeological and Historic Preservation Act

The Archeological and Historic Preservation Act is also not currently an ARAR since no archaeologic or historic sites have been identified at the operable unit. This act is similar to the National Historic Preservation Act but differs in that it mandates only protection of historic or archaeologic data and not the actual archaeologic or historical site.

The Endangered Species Act

The Endangered Species Act of 1973 is not an ARAR since no endangered or threatened species or habitat was identified in the operable unit during ecological surveys of the Hanford Site.

Fish and Wildlife Coordination Act

Requirements under the Fish and Wildlife Coordination Acts are not an ARAR to remedial actions at the site since remedial actions are not expected to involve loss of habitat.

The Wild and Scenic River Act

The Wild and Scenic River Act is not an ARAR since the Columbia River is not currently included in the national system of wild and scenic rivers. The Columbia River has been nominated for inclusion on the list and remedial alternatives proposed for the site should consider migration of contaminants from soils to groundwater and potential impacts to the river. This will be reevaluated as a potential ARAR in the 200 Area Aggregate Management Study for groundwater.

7.2.1.2.2 State Location-Specific ARARs. State of Washington Department of Game procedures for compliance with the Washington State Environmental Policy Act are not an ARAR since threatened, endangered or sensitive wildlife or habitat have not been identified at the operable unit. The act requires that management plans be developed if threatened, endangered, or sensitive wildlife or habitat are affected by remedial actions at the site.

7.2.2 Recommended Expedited Response Actions (ERA)

At the time of the completion of the first phase of the RI, it is appropriate to conduct an NCP expedited response action (ERA) evaluation in accordance with 40 CFR 300.410. This RI report is used, in part, to document such an evaluation for the 200-BP-1 operable unit. Taking factors specified in 40 CFR 300.415(b) (2) into account, this section is used to recommend appropriate removal actions consistent with the guidelines provided in 40 CFR 300.415(d).

An evaluation of the NCP guidelines referenced above indicates that expedited response actions at the 200-BP-1 operable unit are not necessary at this time. There is no evidence of imminent and substantial endangerment to either human health or the environment from exposures to the 200-BP-1 operable unit contaminants (Section 6.2), no sensitive ecosystems are known to be contaminated by the 200-BP-1 operable unit (Section 3.7.2 and 6.3), and none of the contaminants of concern are present in the form of bulk storage (Section 4.1). Although there is some high level soil contamination present, the contaminants of concern are not readily subject to migration due to the interim stabilization activities already conducted over the crib area. Nor are weather conditions expected to arise that would result in the migration of such contamination. There is also no known threat of fire or explosion, and it appears that the 200-BP-1 RI/FS, and any subsequent necessary remediation, can be completed in a timely manner. There is currently no imminent threat from impacted groundwater since groundwater is not extracted for supply in the 200 Areas hydraulically downgradient of the 200-BP-1 operable unit (Section 3.7). Furthermore, there is currently no imminent threat of impact to existing surface water drinking supplies since it does not appear that any plumes originating in the 200-BP-1 operable unit have migrated (Appendix K) to the Columbia River.

Therefore, it is recommended that no expedited response actions be implemented at this time for any of the 200-BP-1 operable unit waste management units. It is recommended, however, that this evaluation be repeated at other appropriate points in the remedial response process.

7.2.3 Recommended Remedial Action Objectives for the Feasibility Study

The initial task of an FS is to develop remedial action objectives. Such objectives establish site remediation goals by taking specific contaminants, contaminated environmental media, and potential contaminant exposure pathways into account (40 CFR 300.430(e) (i)). Remedial action objectives, which are subject to refinement throughout the FS, focus the development, screening, and analysis of remedial alternatives to ensure that they are protective of human health and the environment.

The 200-BP-1 operable unit is a source-operable unit. Groundwater beneath the 200-BP-1 operable unit is included in a separate groundwater-operable unit, the 200 East Groundwater Aggregate Area. An assessment of environmental risk which is posed by the groundwater pathway, and consideration of remedial objectives aimed at addressing groundwater contamination will be included in the 200 East Groundwater Aggregate Area Study.

The existence of a separate operable unit for groundwater in the 200 East Area leads to questions regarding the allocation (separation) and potential overlap of remedial action objectives between the groundwater- and source-operable units. These questions are related primarily to whether, in spite of the fact that groundwater is not included as a part of the source-operable unit, remedial action objectives for source-operable units should address the issue of potential future impacts to groundwater. There is the opportunity, in the separation of groundwater from the source-operable units, for the establishment of a framework which allows for the efficient coordination of remedial actions between the source- and groundwater-operable units in the separations area. This framework is described below.

There are numerous sources of contamination which impact groundwater in the 200 East Area. Impacts from these sources will need to be considered in the remedial plan of the 200 East Groundwater Aggregate study. Impacts from these sources can be addressed in one of the following two ways:

- Groundwater-operable unit studies could include evaluations of impacts to groundwater for each source-operable unit, with the ultimate remedial solution for groundwater dependent on the completion and integration of these individual source studies

- The concept of source control can be adopted for the 200 East Area where it is assumed that, by definition, the individual source-operable unit remedial actions will control source migration to groundwater, where this is likely to occur at unacceptable concentrations. In this manner, the remedial objectives of the groundwater-operable unit RI/FS can be focused on addressing the existing groundwater contamination only, thereby decoupling the groundwater-operable unit study from the individual source-operable unit evaluations.

The second of these two alternatives, by eliminating the dependency of the groundwater study on the source studies, will allow the groundwater-operable unit RI/FS to proceed more quickly and efficiently because of the more limited and clearly defined scope. The work which has been performed in this RI, by evaluating the potential for future contaminant migration to the water table, has been completed in this manner.

Therefore, based on the results of the Fate and Transport Analysis (Chapter 5) and the Baseline Risk Assessment (Chapter 6), and within the context of the approach outlined above, recommended remedial action objectives, which are subject to refinement later in the RI/FS process, include the following:

- **Limit human receptor exposure to near surface and subsurface infiltration gravels/soils.** The results of the risk assessment performed in Chapter 6 indicate that, amongst the various human environmental exposure scenarios which were evaluated, a cancer risk of 10^{-6} is exceeded only under the future industrial scenario on the operable unit, and only if near surface soils or infiltration gravels/soils are exposed, permitting direct contact with the contaminants. As long as the current clean soil cover is maintained, or subsurface soils are not disturbed, the risks to human health will remain below a 10^{-6} cancer risk level. Leaving contaminated soils in-place can be an acceptable remedial approach because no ARARs are exceeded for the near

surface contaminated soils. Near surface UCLs for contaminants of potential concern do not exceed MTCA Method C (industrial scenario) or even Method B (residential scenario) cleanup standards.

- **Limit biotic exposures to near surface and subsurface infiltration gravels/soils.** The results of the ecological risk assessment indicate that biota could be adversely impacted by near surface soil contaminants on the operable unit if exposure to these soils is allowed to occur. A remedial action objective of the FS therefore shall be to include measures which limit such exposures. These measures may largely overlap with and consist of those which will limit human exposures; however, additional measures may also be required to address those pathways which are unique to biota, such as burrowing or systemic uptake by roots.

- **Limit future impacts to groundwater by taking measures which minimize downward migration of contaminants within the vadose zone.** The 200-BP-1 operable unit is currently impacting the underlying unconfined aquifer with Tc-99, nitrate, Co-60 and possibly uranium. The concentration of Tc-99 averages about 400 to 700 pCi/L in groundwater immediately downgradient of the cribs. This average concentration is below the current MCL of 900 pCi/L (note that Tc-99 was at 970 and 990 pCi/L in groundwater from wells 299-E33-5 and 7, respectively, during one sampling period) and well below the proposed MCL of 3800 pCi/L, but slightly above the MCTA Method C standard of 350 pCi/L and over an order of magnitude greater than the MCTA Method B standard of 35 pCi/L. Nitrate contamination of the aquifer appears to be contributed from the operable unit by 10 to 20 mg/L over upgradient groundwater. This contribution is less than the current MCL of 45 mg/L, but contributes to groundwater being above the MCL in much of the operable unit. Current Co-60 and uranium (if from the 200-BP-1 operable unit) impacts to the aquifer are below MCLs, but above MCTA Method B standards. MCTA Method C standards for uranium and Co-60 are not currently exceeded in the aquifer beneath the operable unit.

  In the future, the 200-BP-1 operable unit sources will continue to impact the underlying unconfined aquifer. Impact from Tc-99, nitrate and Co-60 will decrease with time, although the length of time is uncertain. Co-60 will decrease at a more rapid rate due to its relatively short half life of 5 years. Uranium and plutonium are expected to significantly impact the aquifer in the distant future. Uranium could exceed proposed MCLs in groundwater in hundreds-of-years to possibly even a thousand years. Concentrations of uranium from this operable unit could exceed 1000 pCi/L, but is anticipated not to occur for over a thousand years. Plutonium could migrate to the water table in tens-of-thousands of years through primarily diffusive transport mechanisms, and exceed the current MCL of 15 pCi/L and the proposed MCL of 65 pCi/L.

  Remedial action objectives for groundwater impact should be consistent with current groundwater conditions and address future impacts within reasonable time frames. Tc-99, nitrate and C0-60 have already impacted the aquifer in much of the study area and are generally found at much higher

concentrations downgradient of the 200-BP-1 operable unit. The 200 East Aggregate Area Groundwater Study will define remedial action objectives for these plumes and incorporate appropriate measures for the entire 200 East Area. Source control measures do not appear warranted for Tc-99, nitrate and Co-60 since their current impacts from the operable unit are declining with time and are below MCL standards; although they are above MCTA (especially Method B) standards.

On the other hand, remedial action objectives for uranium and plutonium should consider the time frame required for significant impacts from these contaminants. Uranium may currently be entering and impacting the aquifer from the 200-BP-1 operable unit over the MCTA level B standard (1.6 pCi/L) but will not exceed the proposed MCL of 30 pCi/L for hundreds or possibly a thousand years. It is noteworthy that the MCTA Method B standard is below the Hanford Site background of 3.4 pCi/L for uranium. Plutonium is predicted to take 10,000 years to begin to reach the water table.

7.2.4 Recommended Additional Investigative Activities

For additional investigations, and in support of the FS, the following data collection activity is proposed:

- At this time, based upon the remedial action objectives which were outlined above, it is considered likely that remedial action at the 200-BP-1 operable unit could involve the use of a surface barrier or some other encapsulation or immobilization technology. A major cost factor related to the chosen remedial approach will be the physical size (areal extent) of the contaminated crib soils. Currently, there is uncertainty associated with the lateral extent of subsurface contamination. Data collected in the Phase I RI during the spectral gamma-ray logging are limited, but suggest that contamination in the vadose zone may extend beyond 15 m (50 ft) laterally away from the immediate crib area. There is no other information which would assist in providing more definitive data with regards to the true lateral extent of contamination from the crib area. The costs of potential remedial measures are dependent on the lateral extent of contaminants.

  Rather than proposing a program of drilling and soil sampling, it is recommended that plans be developed to utilize the spectral gamma ray logging system (RLS) which was used in this RI. This system can be used in conjunction with the cone penetrometer or in-situ characterization probes which have been employed recently at the Hanford Site. The probes will be advanced to depths of 40 to 50 feet (below maximum zones of contamination) thereby eliminating the generation of any contaminated cuttings. In this manner, the area can be surveyed relatively quickly to establish the limits of the soil contamination. It is proposed that the logging be performed along a series of transects which originate in the immediate vicinity of the cribs. The boreholes should be spaced at regular intervals and emanate outward from the crib area until the extent of the zone of contamination is defined. At this

time, one transect from each side of the crib area is suggested. This data can be obtained during remedial design refinement.

In addition, groundwater sampling and analysis for uranium, Tc-99 and Co-60 should continue from wells within the operable unit immediately adjacent to the cribs. This continued monitoring will provide data for assessing whether groundwater impacts below the operable unit sources are changing as predicted.

Table 7-1. Identification of Federal ARARs at the 200-BP-1 Operable Unit.
(Sheet 1 of 4)

Requirements	Applicable, Relevant & Appropriate, or To Be Considered,	Comment
CHEMICAL SPECIFIC		
Safe Drinking Water Act of 1974 Title 42 USC 300, et seq.		
National Primary Drinking Water Standards 40 CFR 141	Relevant & Appropriate*	The NCP requires that maximum contaminant level goals (MCLGs) and maximum contaminant levels (MCLs) established under the Safe Drinking Water Act be attained by remedial actions for groundwater and surface waters that are current or future sources of drinking water where the MCLG or MCL are relevant and appropriate to the situation. (See Footnote a). Groundwater is currently not used for drinking, however it could be used in the future, if the site is released from institutional controls. In addition, there is potential for discharge of contaminated groundwater to the Columbia River, which is used for drinking water. Remedial alternatives need to prevent migration of contaminants from soils to groundwater at concentrations that cause groundwater to exceed MCLs and MCLGs. MCLs and MCLGs for public drinking water are presented in Tables 7-3 and 7-4 for contaminants of concern.
National Secondary Drinking Water Standards 40 CFR 143	Not ARAR*	Federal secondary standards are not enforceable standards and are not typically applicable or relevant and appropriate requirements. (See Footnote a). Table 7-4 presents secondary drinking water standards for contaminants of concern.
Atomic Energy Act of 1954, as amended Title 42 USC 2011 et seq.		
Environmental Radiation Protection Standards for Nuclear Power Operations 40 CFR 190	Not ARAR	The regulation specifies the levels below which normal operations of the uranium fuel cycle are determined to be environmentally acceptable. These standards are not applicable and not relevant and appropriate since the standard excludes operations at disposal sites and definition of the uranium fuel cycle focuses on those processes that result in generation of electrical power. The standard sets dose equivalents from the facility which are not to exceed 25 mrems/yr to whole body, 75 mrems/yr to thyroid, or 25 mrems/yr to any other organ. Release limits at .5 mCi for Pu-239 and other alpha emitting transuranics with half-lives greater than one year.
Footnote a: WAC 173-340-720 (2)(a)(ii) specifies that MCLs, MCLGs and SMCLs are applicable requirements for groundwater cleanup, where groundwater has a current or potential future use as drinking water.		

Table 7-1. Identification of Federal ARARs at the 200-BP-1 Operable Unit.
(Sheet 2 of 4)

Requirements	Applicable, Relevant & Appropriate, or To Be Considered,	Comment
Environmental Radiation Protection Standards for the Management and Disposal of Spent Nuclear Fuel, High-Level Waste and Transuranic Radioactive Waste 40 CFR Part 191	Applicable	Standards under this regulation are applicable since they contain environmental protection requirements for management and disposal of spent nuclear fuel, high-level waste and transuranic wastes at facilities operated by the Department of Energy. The standard addresses all disposal methods. Subpart A applies to facilities regulated by the Nuclear Regulatory Commission and sets maximum allowable effective dose equivalent for any member of the public at 25 mrems/yr to whole body, 75 mrems/yr to thyroid or 25 mrems/yr to any other organ. Facilities not regulated by the Commission shall provide reasonable assurance that the combined annual dose to the public resulting from discharges shall not exceed 25 mrems/yr to whole body or 75 mrems/yr to any critical organ. Alternative standards may be established that prevent any member of the public from receiving a continuous exposure of more than 100 mrems/yr and an infrequent exposure of 500 mrems/yr from all sources. Environmental standards set in Subpart B address protection of individual members of the public and groundwater at disposal facilities. Disposal systems are to be designed to provide protection for up to 1,000 yr following disposal and undisturbed performance should limit individual members of the public to less than 25 mrem/yr to the whole body or 75 mrems/yr to any critical organ. Radium-226/228 are not to exceed 5pCi/l, alpha emitters are not to exceed 15 pCi/l, including Ra-226/228, excluding radon and combined concentrations of beta or gamma emitters shall not produce an annual dose equivalent to whole body or any internal organ greater than 4 mrem/yr if an individual consumed 2 l/day from the groundwater source. If average annual concentrations in groundwater exceed standards prior to construction of the disposal facility, then the disposal system may not increase the existing conditions by more than the limits set in Subpart B.
Nuclear Regulatory Standards for Protection Against Radiation 10 CFR 20	Relevant & Appropriate	The regulation establishes standards for protection of the public against radiation arising from the use of regulated materials and as such are relevant and appropriate. Radioactive material from sources not licensed by the NRC are not subject to these regulations, therefore this standard is not applicable because the operable unit is not NRC licensed. Remedial alternatives need to limit external and internal exposure from releases to levels that do not exceed 100 mrem/yr, or 2 mrem/ hr from external exposure in unrestricted areas. Specific concentration limits of contaminants of concern in liquid effluent allowed in unrestricted areas are listed in Table 7-3. These limits are based on annual effective dose equivalent from internal exposure for adults of 50 mrem.
Clean Water Act of 1977 33 USC 1251	Applicable	Standards adopted under this act apply to surface water which is not currentiy impacted from the operable unit. Applicability or relevance and appropriateness to groundwater and surface water will be evaluated as part of the groundwater aggregate area management study.

Table 7-1. Identification of Federal ARARs at the 200-BP-1 Operable Unit. (Sheet 3 of 4)

Requirements	Applicable, Relevant & Appropriate, or To Be Considered,	Comment
Clean Air Act of 1977, as amended 42 USC 7401 et seq.		
National Ambient Air Quality Standards 40 CFR 50	Applicable	Requirements of these regulations are applicable to airborne releases of radionuclides and criteria pollutants specified under the statue. Specific release limits for particulates are set at 50 ugm/m^3 annually or 150 ugm/m^3 per 24-hour period. Standards for airborne lead measured as elemental lead are set at 1.5 ugm/m^3, maximum arithmetic mean averaged over a calendar quarter.
National Emission Standard for Hazardous Air Pollutants (NESHAP), Subpart H - National Emission Standards for Emissions of Radionuclides Other than Radon From Department of Energy Facilities 40 CFR 61	Applicable	These requirements are applicable to the site and remedial alternatives since the potential to release air emissions to unrestricted areas exists. Subpart H sets emissions limits from the entire facility to ambient air not exceed an amount that would cause any member of the public to receive an effective dose equivalent of 10 mrem/yr. The definition of facility includes all buildings, structures and operations on one contiguous site.
Resource Conservation and Recovery Act 42 USC 6901 et seq		
Ground Water Protection Standards 40 CFR 264	Not ARAR	Requirements of this section are not applicable or relevant and appropriate since wastes present in soils at the site do not designate as hazardous waste and the 200-BP-1 cribs are not regulated units under RCRA.
Land Disposal Restrictions 40 CFR 268	Not ARAR	Land disposal restrictions are not ARAR to chemicals present at the site since RCRA treatment standards do not concern wastes that are already in place. Land disposal restrictions will be further evaluated as potential action-specific ARARs.
Uranium Mill Tailings Radiation Control Act of 1978 42 USC 2022		
Health and Environmental Protection Standards for Uranium and Thorium Mill Tailings 40 CFR 192	Relevant & Appropriate	Requirements of this act are relevant and appropriate since Ra-226 is present in soils at the site. The standard is not applicable because the operable unit is not a milling site for uranium or thorium. Subpart B sets concentrations limits for Ra-226 in soils for land cleanup of 5 pCi/g averaged over the upper 15 cm and 15 pCi/g averaged over any 15 cm thick layer more than 15 cm from the surface. The level of gamma radiation is any occupiable building is not to exceed 20 microroentgens/hr above background. Groundwater protection requirements for concentrations of Ra-226, Ra-228 and gross alpha particle activity are set at EPA established levels for drinking water.

Table 7-1. Identification of Federal ARARs at the 200-BP-1 Operable Unit.
(Sheet 4 of 4)

Requirements	Applicable, Relevant & Appropriate, or To Be Considered,	Comment
DOE Order 5400.5 - Radiation Protection of the Public and the Environment	To Be Considered	This DOE order sets radiation standards for protection of the public in the vicinity of DOE facilities. The order sets limits for the annual effective dose equivalent of 100 mrem, but allows temporary limits of 500 mrem if avoidance of higher exposures is impractical. The standards sets annual dose limits for any organ at 5 mrem. An annual dose equivalent from drinking water supplies operated by DOE is set at 4 mrem and states that liquid effluent from DOE activities will not cause public drinking water systems to exceed EPA MCLs. Specific concentration limits in water for contaminants of concern are listed in Table 7-3.
Toxic Substance Control Act 15 USC 2601 et seq.		
Regulation of PCBs 40 CFR 761	Not ARAR	TSCA requirements are not ARAR since PCBs detected at the site are below the regulated concentration of 50 ppm. Handling, storage and disposal requirements may require consideration if the 50 ppm levels are exceeded.
LOCATION SPECIFIC		
National Historic Preservation Act of 1966 USC 470 et seq.	Not ARAR	Requirements established under this act are not applicable or relevant and appropriate to the operable unit since no facilities located on the operable unit are currently listed on or proposed for inclusion on the National Register of Historic Places.
Archeological and Historic Preservation Act 16 USC 469a-1	Not ARAR	This act requires that actions conducted at the site must not cause the loss of any archeological and historic data. This act varies from the National Historic Preservation Act in that it mandates only preservation of the data and not the actual facility. No archeological or historic sites have currently been identified within the operable unit and therefore these requirements are not applicable or relevant and appropriate.
Endangered Species Act of 1973 16 USC 1531 et seq.	Not ARAR	Requirements to protect species threatened by extinction and habitats critical to their survival must be considered by remedial alternatives performed on the site. Endangered species and critical habitats have been evaluated at the Hanford Site, and none have been identified in the operable unit, therefore requirements of this act are not applicable or relevant and appropriate.
Wild and Scenic Rivers Act 16 USC 1271 et seq	Not ARAR	Requirements of this act are not applicable or relevant and appropriate since the Columbia River is not included in the national system of wild and scenic rivers. The Columbia River has been proposed for inclusion in the system, and remedial alternatives for the site should consider migration of contaminants from soils to groundwater and potential impact to the Columbia River.

Table 7-2. Identification of State ARARs at the 200-BP-1 Operable Unit.
(Sheet 1 of 2)

REQUIREMENTS	Applicable, Relevant & Appropriate, To be Considered	COMMENT
CHEMICAL SPECIFIC		
Water Pollution Control/ Water Resource Act of 1971 Ch. 90.48 RCW/ Ch.90.54 RCW		
Surface Water Quality Standards WAC 173-201	Applicable	Groundwater below the operable unit is considered to be in communication with surface waters and also discharges to the Columbia River. However, contaminants from the operable unit do not currently discharge to the Columbia River. Potential future impacts will be evaluated in the groundwater aggregate management study. Due to these considerations, requirements of this standard are applicable to the site. Remedial actions must consider migration of contaminants from soils to groundwater and their potential discharge to surface waters.
Hazardous Waste Clean Up/ Model Toxics Control Act Ch. 70.105D RCW		
Model Toxics Control Act WAC 173-340	Applicable	Requirements under this standard are applicable to the operable unit. Specific clean up goals established in the standard require implementation of the strictest federal or state cleanup criteria. For groundwater remediation, MCLs, MCLGs and secondary drinking water standards are identified as cleanup criteria. MTCA also establishes requirements for cleanup of soils based on protection of groundwater. Standards are set at 100 times the most stringent federal or state standard or calculated using methods in the regulation, unless it can be demonstrated this is not appropriate for the site. Vadose zone fate and transport modeling has been used to determine soil clean up levels that are protective for groundwater at 200-BP-1. MTCA surface and groundwater clean-up levels for contaminants of concern are listed in Tables 7-3 and 7-4. MTCA soil clean-up standards for contaminants of concern are also listed in Tables 7-3 and 7-4.
Dangerous Waste Regulations Ch. 70.105 RCW		
Dangerous Waste Regulations WAC 173-303	Not ARAR	Requirements found in the dangerous waste regulations are not applicable or relevant and appropriate to the site since contaminants present do not designate as dangerous waste.

Table 7-2. Identification of State ARARs at the 200-BP-1 Operable Unit.
(Sheet 2 of 2)

REQUIREMENTS	Applicable, Relevant & Appropriate, To be Considered	COMMENT
Solid Waste Management, Recovery and Recycling Act Ch. 70.95 RCW		
Minimum Functional Standards for Solid Waste Handling WAC 173-304	Applicable	The standard is applicable for alternatives requiring management of solid waste. Sets the minimum standards for the handling of all solid waste, including operation, monitoring and closure requirements. The standard sets groundwater maximum contaminant levels (MCLs) at the same levels as the drinking water standards under 40 CFR 141.
State Radiation Standards Ch. 70.98 RCW		
Radiation Protection Standards WAC 246	Relevant & Appropriate	State radiation standards developed pursuant to the Atomic Energy Act of 1954, as amended, are relevant and appropriate to remedial actions conducted at the site. These standards are not applicable since sections of the regulation address specific facility types and activities not associated with the operable unit. Remedial alternatives must not allow maximum annual radiation dose to individuals in restricted areas above 1.25 mrem to the whole body, the head and trunk, active blood forming organs, gonads or lens of the eye; a maximum annual dose of 18.75 mrems to the hands and forearms or feet and ankles; a maximum annual dose of 7.5 mrems to the skin of the whole body. Approval may be granted for alternate dose levels of 3 mrem/yr to the whole body, or a total accumulated dose not to exceed 5 times the individuals age minus 18. Section 246-221-050 requirements set a maximum allowable exposure that minors may receive in restricted areas at a 2mrem/yr permissible level of radiation from external sources in unrestricted areas, or if the individual were continuously present would not result in excess of 100 mrem/ seven consecutive days. Anyone may apply to the agency for approval of an alternate dose as long as the annual dose to an individual does not exceed .5 rem/yr. Section 246-221-070 establishes concentration limits in effluent released to unrestricted areas. Table 7-3 presents the limits for operable unit radionuclides of concern. Section 246-252, Radiation Protection at Uranium and Thorium Milling Operations, contains specific concentration limits for protection of groundwater; gross alpha excluding radon and uranium not to exceed 15 pCi/l, and combined radium-226 and radium-228 not to exceed 5 pCi/l.
LOCATION SPECIFIC		
Department of Game SEPA Procedures WAC 232-12	Not ARAR	Requirements which define actions the Department of Game must take to protect endangered, threatened or sensitive wildlife and habitat. These requirements are not ARAR since no threatened, endangered or sensitive wildlife or habitat has been identified at the operable unit.

Table 7-3. Preliminary Chemical-Specific ARARs for the 200-BP-1 Operable Unit Contaminants of Potential Concern (Radionuclides). (Sheet 1 of 3)

Contaminant	Drinking Water 40 CFR 141[a]	Washington State Model Toxics Control Act Cleanup Regulation[b]				NRC Standards 10 CFR 20[c,d]		Atomic Energy Act, Protection of the Public and Environment, DOE Order 5400.5[e]	
	MCL/ Proposed MCL (pCi/l)	Method B		Method C		Water (pCi/l)	Air (pCi/m^3)	Water (pCi/l)	Air (pCi/m^3)
		GroundWater (pCi/l)	Soil (pCi/g)	GroundWater (pCi/l)	Soil (pCi/g)				
K-40	-	4.2	209	42	8273	-	-	280	36
Sr-90	8[f]/42[g]	1.3	64	13	2528	300/40000	30/200	40	2[h]
Ce-141	300[f]/1890[g]	55.4	2771	554	109640	90000/ 90000	20,000/ 5,000	2000	40[i]
Ce-144	-/261[g]	7.5	377	75	14918	10000/ 10000	300/200	280	1.2
Cs-137	200[f]/119[g]	1.6	82	16	3250	20000/ 40000	2,000/500	120	16[h]
Co-60	100[f]/218[g]	3.1	153	31	6067	50000/ 30000	1,000/100	200	16[h]
Pu-238	15/7.1[g]	.21	10.5	2.1	414	5000/30000	.07/1	1.6	.0012[i]
Pu-239	15/65[g]	.2	10	2.0	396	5000/30000	.06/1	1.2	.0008[i]
Pu-239+240	15/65[g]	.2	10	2.0	396	5000/30000	.06/1	1.2	.0008[i]
Total U	-/30[g]	1.6[j]	82	16[j]	3250	40000/4000 0	3/5	24	.08[i]
Ra-226	5/20[g]	.38	19	3.8	758	30/3000	3/2	4	.004[i]
Sb-125	300[f]/1940[g]	55	2738	550	108330	100000/ 100000	20,000/ 900	2000	40
Th-228	15[f]/153[g]	.84	42	8.4	1655	7000/ 10000	.3/.2	16	.002[i]
Tc-99	900[f]/3,790[g]	35	1769	350	70000	300000/ 200000	70,000/ 2,000	4000	80[i]

Table 7-3. Preliminary Chemical-Specific ARARs for the 200-BP-1 Operable Unit Contaminants of PotentialConcern (Radionuclides). (Sheet 2 of 3)

Contaminant	Drinking Water 40 CFR 141[a]	Washington State Model Toxics Control Act Cleanup Regulation[b]				NRC Standards 10 CFR 20[c,d]		Atomic Energy Act, Protection of the Public and Environment, DOE Order 5400.5[e]	
	MCL/ Proposed MCL (pCi/l)	Method B		Method C		Water (pCi/l)	Air (pCi/m^3)	Water (pCi/l)	Air (pCi/m^3)
		GroundWater (pCi/l)	Soil (pCi/g)	GroundWater (pCi/l)	Soil (pCi/g)				
H-3	20,000[f]/ 60,900[g]	850	42593	8500	1685200	3000000/ 3000000	200000/ 200000	80000	4,000
Gross Alpha	15[f]/15[g]	-	-	-	-	-	-	-	-
Gross Beta	4 mrem/yr	-	-	-	-	-	-	-	-

[a] - State Drinking Water Standards, WAC 246-290, are as stringent as current federal MCLs, unless otherwise noted
[b] - Calculated using the formula in WAC 173-340.
[c] - Appendix B, Table II, Column 2, Concentration Limits for Radionuclides in Liquid Effluent Released to Unrestricted Areas (soluble/insoluble)
[d] - Water Quality Standards for the State of Washington (WAC 173-201) concentrations for all classes of radionuclides are determined by 1/100th of the value listed in WAC 246-221, Appendix A, Table II, Column 2, which is an equivalent to 10 CFR 20, Appendix B, Table II, Column 2
[e] - Reported as 4% of the Derived Concentration Guide for Ingested Water
[f] - Current MCL
[g] - Proposed MCL as reported in the Advanced Notice of Proposed Rule published in 56 FR 33050, July 18, 1991. The notice also published a proposed MCLG of 0 for the radionuclide. MCLGs have not been promulgated for radionuclides.
[h] - ICRP Lung Retention Class W
[i] - ICRP Lung Retention Class Y
[j] - Total U calculated as U-235
- Criteria not listed

Table 7-3. Preliminary Chemical-Specific ARARS for the 200-BP-1 Operable Unit Contaminants of Potential Concern (Radionuclides).
(Sheet 3 of 3)

DOSE RELATED STANDARDS FOR RADIONUCLIDES
Environmental Standards for the Uranium Fuel Cycle 40 CFR 190 The annual dose equivalent is not to exceed 25 mrems/yr to whole body 75 mrems/yr to thyroid 25 mrems/yr to any other organ
Environmental Radiation Protection Standards for Management and Disposal of Spent Nuclear Fuel, High-Level and Transuranic Radioactive Wastes 40 CFR 191 Subpart A - Facilities regulated by the commission are not to exceed standards set by 40 CFR 190. Facilities not regulated by the Commission shall provide reasonable assure that the combined annual dose to the public resulting from discharges shall not exceed 25 mrems/yr to whole body 75 mrems/yr to any critical organ Alternative standards may be established that prevent any member of the public from receiving continuous exposure of more than 100 mrems/yr and an infrequent exposure of 500 mrems/yr from all sources. Subpart B - Environmental Standards for Disposal sets groundwater protection levels at Radium-226/228 at 5 pCi/l Alpha emitters at 15 pCi/l, including Ra-226/228, excluding Radon Combined concentrations of beta or gamma emitters shall not produce an annual dose equivalent to whole body or any internal organ greater than 4 mrem/yr if an individual consumed 2 l/day from the groundwater source
National Emission Standards for Hazardous Air Pollutants 40 CFR 61 Subpart I - National Emission Standards for Radionuclide Emissions from Facilities Licensed by the Nuclear Regulatory Agency and Federal Facilities not Covered by Subpart H Emission of radionuclides, including iodine, to the ambient air from a facility regulated under this subpart shall not exceed those amounts which cause any member of the public to receive in any year an effective dose equivalent of 10 mrem/yr

Table 7-4. Preliminary Chemical-Specific ARARs for the 200-BP1 Operable Unit Contaminants of Potential Concern (Non-Radioactive Contaminants)
(Sheet 1 of 2)

Contaminant	Drinking Water Standards 40 CFR 141[a] and 40 CFR 143[b]		Washington State Model Toxics Cleanup Act WAC 173-340				
	MCLs	MCLGs	Method B		Method C		
	current/ proposed	current/ proposed	Ground Water 173-340-720	Soil 173-340-740	Ground Water 173-340-720	Soil 173-340-740	Soil 173-340-745
	(mg/l)	(mg/l)	(mg/l)	(mg/kg)	(mg/l)	(mg/kg)	(mg/kg)
Antimony	-/.006[e]	-/.006[e]	.006[e]	32[d]	.006[e]	128[d]	1400[d]
Arsenic	.05	-	.005[f]	.59[d]	.005[f]	23[d]	76[d]
Barium	1 / 2[f]	- / 2[f]	1[e]	5600[d]	1[e]	22400[d]	2.5E[d]
Cadmium	.005	.005	.005[e]	80[d]	.005[e]	320[d]	3500[d]
Chromium	.1	.1	.1[e]	400[d]	.1[e]	1600[d]	17500[d]
Copper	1[b] / 1.3[g]	-	.64	3200	1[b]	12800	1.4E5[d]
Lead	.05 / .015[g]	-	.05[e]	-	.05[e]	-	-
Manganese	.05[b]	-	1.6[d]	8000[d]	3.5[d]	32000[d]	3.5E5[d]
Nickel	- / .1[h]	- / .1[h]	.32[d]	1600[d]	.7[d]	6400[d]	7000[d]
Selenium	.05	.05	.05[e]	400[d]	.05[e]	1600[d]	17500[d]
Silver	.05 / .1[b]	-	.05[e]	400[d]	.05[e]	1600[d]	17500[d]
Thallium	- / .002[f]	- / .0005[f]	.0014[d]	7.2[d]	.002[e]	29[d]	315[d]
Vanadium	-	-	.1[d]	560[d]	.2[d]	2240[d]	24500[d]
Cyanide (total)	-/.2[e]	-/.2[e]	.32[f]	1600[f]	.7[f]	6400[f]	70000[f]
Cyanide (free)	-/.2[e]	-/.2[e]	.32[d]	1600[d]	.7[d]	6400[d]	70000[d]
Cyanide (complex)	-/.2[e]	-/.2[e]	.32[f]	1600[f]	.7[f]	6400[f]	70000[d]
PCBs	.0005	0	1.1E-5[d]	.13[d]	1.1E-4[d]	5.2[d]	17[d]
Tributyl phosphate	-	-	.08[h]	400[h]	.175[h]	1600[h]	17500[h]
4,4"-DDT	-	-	2.6E-4[d]	2.9[d]	2.6E-3[d]	.12[d]	382[d]
Trichloroethene	.005	0	.005[e]	91[d]	.005[e]	3636[d]	12000
Fluoride	4	4	.96[d]	48,000[d]	2.1[d]	19200[d]	2.1E05[d]
Nitrate	44	44	26[d]	1.28E05[d]	44[e]	5.12E05[d]	5.6E06[d]

Table 7-4. Preliminary Chemical-Specific ARARs for the 200-BP-1 Operable Unit Contaminants of Potential Concern (Non-Radioactive Contaminants)
(Sheet 2 of 2)

Contaminant	Drinking Water Standards 40 CFR 141[a] and 40 CFR 143[b]		Washington State Model Toxics Cleanup Act WAC 173-340				
Nitrite	33	33	1.6[d]	8000[d]	3.5[d]	32000[d]	3.5E05[d]
Sulphate	250[b]	-	-	-	-	-	-

[a] - State MCLs and MCLGs are based on federal standards, as amended.
[b] - Secondary Drinking Water Standard under 40 CFR 143.
[c] - MCLs as noted in final rule published July 1992, effective January 17, 1994.
[d] - Reference doses and carcinogenic slope factors taken from IRIS, EPA 1992b.
[e] - MTCA requires cleanup concentrations to be as stringent as applicable state or federal standards. MCLs are reported since MTCA Method B and C calculated cleanup concentrations exceed state and federal drinking water MCLs.
[f] - Cleanup level based on background concentration for the State of Washington as noted in Table 1, footnote b, WAC 173-340-720.
[g] - Effective date December 7, 1992, as published June 7, 1991.
[h] - Proposed MCL, effective January 1, 1993.
[i] - Based on toxicity factor for free cyanide.
[j] - Calculated using RfD for PAHs, IRIS, EPA 1992b.
[k] - Reference dose supplied by the Superfund Technical Support Center (STSC, EPA 1992c).
- Criteria not listed.

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